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Long-Term Effects of Land Application of Domestic Wastewater, Hollister, California, Rapid Infiltration Site

Metcalf and Eddy, Inc, Palo Alto, Calif

Prepared for

Robert S Kerr Environmental Research Lab, Ada, Okla

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LONG-TERM EFFECTS OF LAND APPLICATION OF DOMESTIC WASTEWATER: Hollister, California, Rapid Infiltration Site

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

Robert S. Kerr Environmental Research Laboratory
Office of Research and Development
U.S. Environmental Protection Agency
Ada, Oklahoma 74820

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Primary municipal effluent has been applied continuously to rapid infiltration basins at Hollister, California, for more than 30 years. The current daily flow is 43.8 L/s (1.0 Mgal/d). Annual wastewater application equals 15.4 m (51 ft) to 20 infiltration basins intermittently flooded for 1 to 2 days every 14 to 21 days, depending on basin size and season of year.

Infiltration rates were determined, subsurface hydrology was logged, and water table response to wastewater application was monitored. A sampling and analysis program covering a 1 year period included samples from (1) primary effluent, (2) onsite and control site soil profiles, (3) groundwater at the site and upgradient and downgradient of groundwater movement from the site.

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Hollister, California, Rapid Infiltration Site

by

Charles E. Pound
Ronald W. Crites
James V. Olson
Metcalf & Eddy, Inc.
Palo Alto, California 94303

Contract No. 68-03-2361

Project Officer

William R. Duffer, Ph.D.
Wastewater Management Branch
Robert S. Kerr Environmental Research Laboratory
Ada, Oklahoma 74820

ROBERT S. KERR ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
ADA, OKLAHOMA 74820

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FOREWORD

The Environmental Protection Agency was established to coordinate the administration of major Federal programs designed to protect the quality of our environment.

An important part of the agency's effort involves the search for information about environmental problems, management techniques, and new technologies through which optimum use of the nation's land and water resources can be assured and the threat pollution poses to the welfare of the American people can be minimized.

EPA's Office of Research and Development conducts this search through a nationwide network of research facilities. As one of these facilities, the Robert S. Kerr Environmental Research Laboratory is responsible for the management of programs including the development and demonstration of soil and other natural systems for the treatment and management of municipal wastewaters.

Although land application of municipal wastewaters has been practiced for years, there has been a growing and widespread interest in this practice in recent years. The use of land application received major impetus with the passage of the 1972 amendments to the Federal Water Pollution Control Act. The 1977 amendments to the Act gave further encouragement to the use of land application and provided certain incentives for the funding of these systems through the construction grants program. With the widespread implementation of land application systems, there is an urgent need for answers to several major questions. One of these questions regards the long-term effects of land application on the soil, crops, groundwater, and other environmental components. This report is one in a series of ten which documents the effects of long-term wastewater application at selected irrigation and rapid infiltration study sites. These case studies should provide new insight into the long-term effects of land application of municipal wastewaters.

This report contributes to the knowledge which is essential for the EPA to meet the requirements of environmental laws and enforce pollution control standards which are reasonable, cost effective, and provide adequate protection for the American public.

William C Galegar

William C. Galegar
Director
Robert S. Kerr Environmental Research Laboratory

ABSTRACT

The objective of this study was to evaluate the long-term effects of applying municipal wastewater after primary treatment to the land using the rapid infiltration technique. This was accomplished by analyzing groundwater quality and soil chemistry at a site with a long operating history.

Primary municipal effluent has been applied continuously to rapid infiltration basins at Hollister, California, for more than 30 years. The current daily flow is 43.8 L/s (1.0 Mgal/d). Annual wastewater application equals 15.4 m (51 ft) to 20 infiltration basins intermittently flooded for 1 to 2 days every 14 to 21 days, depending on basin size and season of year.

Infiltration rates were determined, subsurface hydrology was logged, and water table response to wastewater application was monitored. A sampling and analysis program covering a 1 year period included samples from (1) primary effluent, (2) onsite and control site soil profiles, and (3) groundwater at the site and upgradient and downgradient of groundwater movement from the site.

Analysis of the primary effluent and groundwater results indicated that COD, BOD, TOC, nitrogen, and fecal coliform bacteria were effectively reduced after percolation through 7 m (22 ft) of unsaturated gravelly and sandy loam soil. Effective phosphorus removal required longer travel distances but the sorption capacity of the soil has not been exceeded after 30 years of continuous wastewater application. Trace element retention by the soil was low; however, only lead exceeded EPA drinking water limits in the wastewater and shallow groundwater aquifer. Iron and manganese are both being leached from the soil with the percolating wastewater. Only slight boron removal in the percolate was observed.

This report was submitted in fulfillment of Contract No. 68-03-2361 by Metcalf & Eddy, Inc., under the sponsorship of the U.S. Environmental Protection Agency. This report covers the period January 2, 1976, to September 2, 1977, and work was completed as of December 2, 1977.

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LIST OF ABBREVIATIONS

Special

ABS	Alkylbenzenesulfonate
ANOVA	Analysis of variance
BMDP2V	Computer program designation to generate two-way analysis of variance
BOD	Biochemical oxygen demand
CCE	Carbon chloriform extract
CEC	Cation exchange capacity
COD	Chemical oxygen demand
DTPA	Diethylenetriamine pentacetic acid
EC	Electrical conductivity
E. Coli	*Escherichia coli*
EPA	Environmental Protection Agency
EPR	Exchangeable potassium ratio
ESR	Exchangeable sodium ratio
JTU	Jackson turbidity units
LAS	Linear alkyl benzene sulfanate
M	Molarity
N	Normality
Nm	Not measurable
PAR	Potassium adsorption ratio
PVC	Polyvinylchloride
SAR	Sodium adsorption ratio
SPSS	Statistical package for the social sciences
SS	Suspended solids
TDS	Total dissolved solids
TEA	Triethanolamine
TKN	Total Kjeldahl nitrogen
TOC	Total organic carbon
USPHS	United States Public Health Service

Quantity

kg/ha	Kilogram per hectare
lb/acre	Pound per acre
mg/kg	Milligram per kilogram
meq/100 g	Milliequivalents per 100 grams
meq/L	Milliequivalents per litre
μg/g	Micrograms per gram
μmhos/cm	Micromhos per centimeter
ppm	Parts per million
g/cm^3	grams per cubic centimeter

LIST OF SYMBOLS

Miscellaneous

^{o}C	Degrees Celsius
ρ	Soil particle density
μ	True mean of a subscripted soil variable
α	Level of significance attached to statistical test; the probability of rejecting a true hypothesis; also a multiple regression constant
%	Percent
%P	Percent pore volume
$\partial s/\partial t$	Partial differential of sorbed phosphorus with respect to time
2.5 Y 5/2	Soil Conservation Service (SCS) soil color symbol
β,m,a,b,d	Multiple regression constants
A	Soil surface area (Eq. 1); also shallow well designation
Ap, A1 C1, C2	Soil morphogenic master horizon designations
$[A^{+}]$, $[B^{+}]$	Chemical activity of solution ions
(Ax), (Bx)	Exchanger concentrations
B	Intermediate well
C	Deep well
C:N	Carbon to nitrogen ratio

D_1, D_2, D_3, D_4	Depth increments 1 through 4
D_b	Soil bulk density
Eh	Oxidation-reduction potential
F	Variance ratio
K_{ab}	Selectivity coefficient
k,n	Constants related to energy of sorption (Eq. 4); also multiple regression constants
H_o	Hypothesis
M	Mass of soil
P	Probability level relative to predetermined "α" value; also phosphorus
r	Soil particle radius
r^2	Correlation coefficient squared
S	Sorbed phosphorus
t	Statistic for testing difference between means
T_1	Control site statistical treatments
T_2	Application site statistical treatment
X	Observation

Mathematical

±	Plus or minus
>	Greater than
<	Less than
1:1	1 to 1 ratio

LIST OF SYMBOLS (Continued)

Elements

Ag	Silver
As	Arsenic
B	Boron
Ba	Barium
Ca	Calcium
Cl	Chlorine
Cd	Cadmium
Co	Cobalt
Cr	Chromium
Cu	Copper
F	Fluorine
Fe	Iron
H	Hydrogen
Hg	Mercury
K	Potassium
Mg	Magnesium
Mn	Manganese
N	Nitrogen
Na	Sodium
Ni	Nickel
P	Phosphorus
Pb	Lead
Se	Selenium
Zn	Zinc

Compounds

$CaCO_3$	Calcium carbonate
CaF_2	Calcium fluoride
$CaNO_3$	Calcium nitrate
$Ca_5(PO_4)_3F$	Fluorapatite
H_2O	Water
HCL	Hydrochloric acid
H_3BO_3	Boric acid
$HCLO_4$	Perchloric acid
$K_2Cr_2O_7$	Potassium dichromate
$MgHCO_3$	Magnesium bicarbonate
MnO_2	Manganese dioxide
$NaCO_3$	Sodium carbonate
$NaHCO_3$	Sodium bicarbonate
NH_3-N	Ammonia-Nitrogen
NH_4^+	Ammonium ion
NH_4OAc	Ammonium acetate
N_2	Nitrogen gas
N_2O	Nitrous oxide
NO_2-N	Nitrite nitrogen
NO_3-N	Nitrate nitrogen
O_2	Oxygen gas
PO_4^{-3}	Phosphate
SO_4^{-2}	Sulfate

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SECTION 1

INTRODUCTION

BACKGROUND

Land treatment processes for renovating wastewater have been demonstrated successfully on many occasions in the past, and in the last 40 years the number of land treatment facilities has increased steadily in the United States [1, 2]. Land treatment has demonstrated equal or superior pollutant removal when compared to conventional treatment and, in many cases, beneficial water reuse is an integral part of the process. In the future, as construction costs increase and water resources become more scarce, land treatment will become even more competitive as a wastewater management alternative.

Land treatment of municipal wastewater encompasses a wide variety of processes or methods. The three principal processes, are:

1. Slow rate
2. Rapid infiltration
3. Overland flow

The major concepts involved in these processes are defined herein.

The term slow rate land treatment is used to focus attention on wastewater treatment rather than on irrigation of crops. However, in slow rate systems, vegetation is a critical component for managing water and nutrients. The applied wastewater is treated as it flows through the soil matrix, and a portion of the flow percolates to the groundwater. Surface runoff of the applied water is generally not allowed.

In rapid infiltration land treatment (referred to in previous U.S. Environmental Protection Agency (EPA) reports as infiltration-percolation), most of the applied wastewater percolates through the soil, and the treated effluent eventually reaches the groundwater. The wastewater is applied to rapidly permeable soils, such as sands and loamy sands, by spreading in basins or by sprinkling, and is treated as it travels through the soil matrix. Vegetation is not usually used, but there are some exceptions. In many cases, recovery of renovated water is an integral part of the system. This can be accomplished using underdrains or wells.

In overland flow land treatment, wastewater is applied over the upper reaches of sloped terraces and allowed to flow across the vegetated surface to runoff collection ditches. The wastewater is renovated by physical, chemical, and biological means as it flows in a thin film down the relatively impermeable slope.

With the relatively large number of successful land treatment systems currently in operation, a wealth of monitoring data should be available. Most of the existing land treatment systems, however, are inadequately monitored. As a result, there is concern about limitations on the operating life of land treatment systems. Specific questions regarding (1) the level of preapplication treatment necessary; (2) the movement of persistent organics, nitrates, and trace elements into groundwaters; (3) accumulations of salt or toxic materials in soils; or (4) translocation of potentially toxic trace elements from the soil into crops remain to be answered. Answers to these and other related questions will assure that sound design criteria can be established and that land treatment systems can be implemented without risk to public health.

The EPA and others have recently funded research on the environmental effects of different land application techniques in pilot and experimental studies; however, operating land treatment systems with substantial longevity have not been studied or monitored extensively. Complete case studies of the effects of groundwater quality and movement in a number of these systems should provide some answers to the concerns that were mentioned. This report presents the findings of a research study at a 30-year old land treatment system at Hollister, California, where primary municipal effluent is applied to rapid infiltration basins.

OBJECTIVE

The objective of this study was to evaluate the long-term effects on groundwater quality and soil chemistry at a site with a long operating history of applying primary municipal effluent to the land using the rapid infiltration technique. The long-term effects were determined by comparing groundwater quality and soil chemistry data at this site with data from a nearby control site having similar physical characteristics and management except that wastewater has never been applied.

SPECIFIC GOALS

To accomplish the overall objective, the following specific goals were selected:

1. Determine the nature of the site hydrogeology for the purpose of relating subsurface conditions to the efficiency of the soil matrix in removing constituents from the applied primary effluent and, subsequently, the danger of contaminating native groundwater aquifers.

2. Determine the current treatment efficiency of the soil matrix by comparing the quality of the applied primary effluent with the groundwater at the site.

3. Determine the long-term ability of the soil matrix to treat wastewater by calculating the mass of selected constituents retained in the soil profile applied over the 30 year operating life.

SECTION 2

CONCLUSIONS

GENERAL

Effective long-term treatment capability has been demonstrated at a rapid infiltration site where unchlorinated primary effluent from the City of Hollister, California, has been managed successfully for 30 years. Thus, a fully managed groundwater recharge system for renovating primary municipal effluent by surface spreading at high rates should provide excellent treatment for a long period of time.

HYDROGEOLOGY

1. The subsurface hydrology at the rapid infiltration site consists of a complex network of water-bearing and impermeable material deposited as stream sediments. Beneath the site there are probably three water-bearing zones--shallow, less than 12.7 m (42 ft); intermediate, between 15 m to 25 m (45 to 75 ft); and deep, greater than 55 m (165 ft)--each separated by semicontinuous to continuous clay and silt layers.

2. The infiltrating effluent apparently forms a mound caused by perching of the effluent on top of the clay layer that underlies the shallow permeable zone. When infiltration ceases, the mound subsides mainly by lateral outflow from the site.

3. The intermediate wells showed no response to effluent application to the infiltration basins. However, some vertical leakage may be occurring, especially where the clay and silt between the shallow and intermediate zones is the thinnest. This conclusion is based on a comparison of concentrations of some mobile ions in the shallow and intermediate groundwater.

4. The deep observation wells did not show any response to effluent application. The lower permeable zone is overlain by an average of 26 m (85 ft) of clay and silt that is probably continuous beneath the treatment site and is an effective barrier to vertical groundwater movement.

SOIL PHYSICAL PROPERTIES

1. The most appropriate soil mapping terms for the alluvial deposits at the site appear to be gravelly sand or gravelly sandy loam.

2. There were insufficient bulk density measurements to determine if conclusive differences existed between control and treatment sites or among depth increments. Measurements taken ranged from 1.27 to 1.72 g/cm^3. Lower than average bulk densities of the near surface measurements at the treatment sites were probably the result of disking operations.

INFILTRATION RATES

1. The maximum long-term infiltration rate obtainable, based on wastewater intake rates monitored over the study period, was 64.6 m/yr (212 ft/yr).

2. Initial wastewater intake was slowed by clogging of soil pores at the surface from wastewater solids applied during the flooding cycle. However, the initial water intake rate was restored by surface disking between each wastewater application. No significant decrease in soil profile infiltration capacity was indicated after 30 years of wastewater application. This conclusion is based on the results of cylinder infiltrometer tests at an actual treatment site location and a similar offsite location where wastewater has never been applied.

3. The water intake capacity of the treatment site soils did not, on the average, limit the amount of wastewater that could be applied as measured by the time it took for complete infiltration of the applied effluent. However, reflooding a basin without prior surface disking caused a greater degree of soil pore clogging and a subsequent lower infiltration rate.

4. Final infiltration rates measured by cylinder infiltrometers were generally consistent, but overestimated by a factor of 16, the rates obtained by flooding an entire basin. No significant difference between buffered and unbuffered cylinder infiltrometer infiltration rates was observed. The clogging effect of wastewater solids was not observed during cylinder infiltrometer tests comparing primary effluent (SS = 275 mg/L) and clear tap water.

SOIL CHEMISTRY

pH and Calcium Carbonate

The addition of wastewater to the treatment site significantly decreased the soil pH at all depth increments through 300 cm (10 ft). Simultaneous calcium carbonate depletion from the site reduced the buffering capacity of the soil permitting a pH reduction. Surface soil pH was less than applied wastewater pH, suggesting nitrification as an additional mechanism for pH reduction.

Organic Matter

Only a slight buildup of organic matter was observed as a result of wastewater application, thus suggesting that the soil microbial population existed in sufficient numbers to bio-oxidize most of the incoming organic matter.

Nitrogen

Significant concentration increases resulting from wastewater treatment were observed for the nitrogen species measured (total and organic nitrogen) to a depth of 100 cm (39 in.). The greatest accumulation of nitrogen was found at the surface. Only 2% of the total nitrogen applied over 30 years was accounted for in the soil profile. This suggests the conversion of nitrogen to mobile forms and movement from the soil profile either to the atmosphere or the groundwater. Groundwater quality results indicate that 93% of the total nitrogen in the applied wastewater was removed from solution.

Phosphorus

The fact that extractable phosphorus was so much higher at all sampling depths in the treatment site suggests that availability for plant uptake was high and that soil solution phosphorus may have passed through the soil profile. Soil mass balance calculations further suggest significant transport of phosphorus to the underlying groundwater.

Laboratory phosphorus isotherm studies revealed that after 30 years of wastewater application, the soil profile at 0 to 16 cm (0 to 6 in.) is retaining 68% of its experimentally determined sorption capacity. Existing phosphorus sorption models were shown to underestimate the actual 30 year phosphorus sorption capacity by factors of 4 to 15 based on measured phosphorus in Hollister treatment site soils.

Boron

Boron was observed to increase fourfold in treatment site soils with the most extensive accumulation near the surface.

CEC and Exchangeable Cations

1. Wastewater application caused a significant increase in CEC with respect to depth and treatment in spite of the pH decrease which simultaneously occurred. It is therefore presumed that the resulting loss of any pH dependent charge was more than offset by the influx of organic matter and mineral clays in the wastewater.

2. The data indicate that more exchangeable sodium is present than can be predicted by the calculated ESR (exchangeable sodium ratio). One explanation is concentration of sodium on the surface by evaporation before transport through the soil profile. An alternative explanation is that magnesium and calcium are ion-paired, complexed, or chelated to a much greater extent than sodium. The result is a low SAR (sodium adsorption ratio) value and therefore the calculated ESR underestimates the exchange phase sodium.

3. High ESR values throughout the treatment site profile have not caused any apparent decrease in infiltration capacity.

4. The data indicate that magnesium was selectively replacing calcium in the surface 100 cm (39 in.) of the soil. The newly mobilized calcium traveled downward and then appeared to exchange with magnesium at 300 cm (10 ft).

Heavy Metals

1. The increase in extractable iron throughout the soil profile may result from its accumulation by wastewater addition, or from its increased extractability as a result of alternating periods of wetness and dryness. There is no doubt, however, that iron was moving through the soil profile.

2. In contrast to iron, manganese was leached by the wastewater from the surface 30 cm (1 ft) with some redeposition at depth.

3. Nickel and cobalt did not accumulate at the surface 30 cm (1 ft) where manganese was lost. However, extractable cobalt was six times higher in treatment sites at 300 cm (10 ft) than in control sites at the same depth. Similarly, extractable nickel values doubled at the 100 cm (39 in.) depth.

4. Wastewater application caused extractable cadmium to increase significantly at 30 cm (1 ft).

5. The increase in extractable zinc to the 300 cm (10 ft) depth suggests that some wastewater applied zinc has passed down to and perhaps through this depth. The apparently higher mobility of zinc, as compared to chemically similar cadmium, may be related to its higher input concentration in the wastewater.

6. Significant extractable copper showed a pronounced accumulation in the soil profile at 0 to 16 cm (0 to 6 in.).

7. The levels of soil DTPA-extractable metals will not adversely affect the future agricultural potential of the Hollister rapid infiltration site.

GROUNDWATER QUALITY

1. On the basis of existing quality criteria for groundwater, there was no evidence that toxic trace elements or potentially pathogenic bacteria associated with municipal wastewater were entering the regional groundwater supply, at harmful levels, as a result of wastewater application to the land at Hollister, California.

2. Analysis of the wastewater and groundwater results indicates that the soil filtration process has effectively reduced the COD, BOD, TOC, and fecal coliform bacteria. After 30 years of continuous wastewater application, COD, BOD, TOC, and fecal coliforms are reduced by 93, 96, 96, and 99%, respectively, after percolation through 7 m (22 ft) of unsaturated gravelly and sandy loam soil.

3. Almost complete nitrification and denitrification of the renovated wastewater was indicated by the results after percolation to the shallow groundwater. The favorable carbon to nitrogen ratio of 6:1 apparently supplies sufficient energy to sustain denitrifying bacteria. No significant nitrogen increase from land applied wastewater was observed in the intermediate or deep observation wells.

4. Effective phosphorus removal required longer travel distances. Shallow groundwater phosphorus concentrations indicated that 22 to 35% of the wastewater phosphorus is currently being removed after 30 years of wastewater application. This agreed closely with the accumulated mass of phosphorus found in the treatment site and soil profile over the 30 year operating history.

5. Trace element concentrations in the shallow groundwater were generally the same as the applied effluent indicating little removal. However, levels of all toxic trace elements, except lead, were below EPA drinking water standards in the wastewater and shallow groundwater. Wastewater lead concentrations were greater than drinking water limits, but were less than concentrations in shallow observation wells. The indication is that lead is being leached from the soil with the percolating wastewater.

6. Manganese concentrations in the shallow groundwater were greater than wastewater manganese concentrations. Manganese is being leached from the soil with the percolating wastewater.

7. Comparison of wastewater and shallow groundwater iron concentrations strongly suggested that applied iron is highly mobile and moving through the soil profile.

8. Only slight boron removal (14%) was observed after percolation to the shallow groundwater.

9. Fluoride concentrations increased slightly in the shallow groundwater over the applied effluent.

SECTION 3

RECOMMENDATIONS

1. The operating history, system operation, groundwater hydrology, and pollutant removals determined in this study can be used in part to design a rapid infiltration system that will not impair land resources or groundwater resources.

2. Primary treatment should be considered adequate and preferred to secondary level pretreatment for control of nitrogen. The carbon to nitrogen ratio of primary effluent favored complete denitrification and did not inhibit COD and BOD removal after relatively short underground travel distance.

3. A study should be undertaken to determine the soil depth that is necessary to achieve satisfactory wastewater treatment when primary effluent is applied at moderate rates.

4. Monitoring of the Hollister rapid infiltration system should be expanded to further define the preferred pathway of subsurface flow from the site. This would provide the means to determine the underground travel distance necessary to achieve a higher degree of treatment than determined herein. Special emphasis should be given to phosphorus and fecal coliform removals.

5. The greatest void of information remaining with respect to land treatment systems is that of persistent or refractory organic compounds. Uncertainties regarding health effects from transport of these materials through the soil from land applied wastewater must be answered before essential design criteria can be established. Quantification of basic scientific data on organic substances of known or suspected toxicity and determination of safe underground travel distances are major areas where research is needed.

SECTION 4

PROJECT DESCRIPTION

Hollister is located in the San Juan Valley 35 km (22 miles) inland from Monterey Bay, 144 km (90 miles) south of San Francisco (Figure 1). Currently, the population served by sewers in and around Hollister is about 10,000. The rapid infiltration site is about 1.6 km (1 mile) west of the city, 150 m (500 ft) south of the San Benito River bed. A summary of important operating information for the Hollister wastewater management system is shown in Table 1.

Table 1. OPERATING FACTORS, HOLLISTER RAPID INFILTRATION SYSTEM

Preapplication treatment	Primary
Groundwater level[a], m	5.8-9.2
Infiltration area, ha	8.8
Total basins	20
Annual wastewater application, m/yr	15.4
Average daily flow, L/s	43.8
Length of operation, yr	30
Industrial influence	Yes[b]

a. Shallow water table created by infiltrating effluent.

b. Slaughterhouse = 0.9 L/s and paper recycler = 11.0 L/s, about 27% of total flow.

m x 3.281 = ft
ha x 2.471 = acre
L/s x 0.0228 = Mgal/d

The facilities at the Hollister rapid infiltration site are shown in Figure 2. Preapplication treatment consists of primary clarification of the untreated influent wastewater. Sludge from the clarifier is regularly drawn off and stored in a converted Imhoff tank before being applied to sludge drying beds independent of the rapid infiltration area.

A portion of influent wastewater flow is equalized in an excavated earthen reservoir before entering the head works and clarifier. Wastewater is pumped from the equalizing reservoir each day when the flowrate is lowest at 2 a.m.

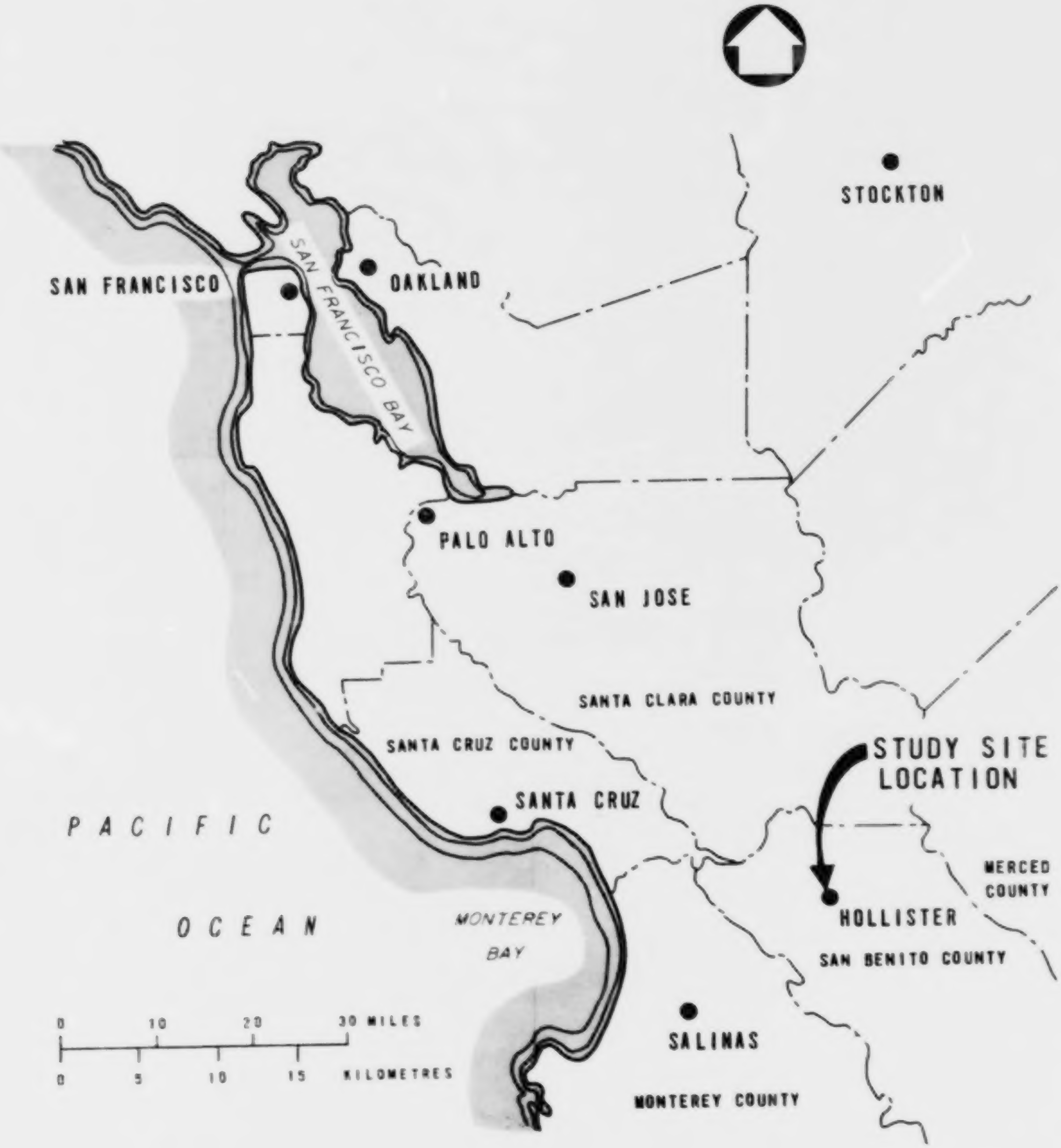
STOCKTON
SAN FRANCISCO
OAKLAND
SAN FRANCISCO BAY
PALO ALTO
SAN JOSE
SANTA CLARA COUNTY
SANTA CRUZ COUNTY
STUDY SITE LOCATION
SANTA CRUZ
PACIFIC
OCEAN
MONTEREY BAY
MERCED COUNTY
HOLLISTER
SAN BENITO COUNTY
SALINAS
MONTEREY COUNTY
0 10 20 30 MILES
0 5 10 15 KILOMETRES

Figure 1. Location of study site Hollister, California.

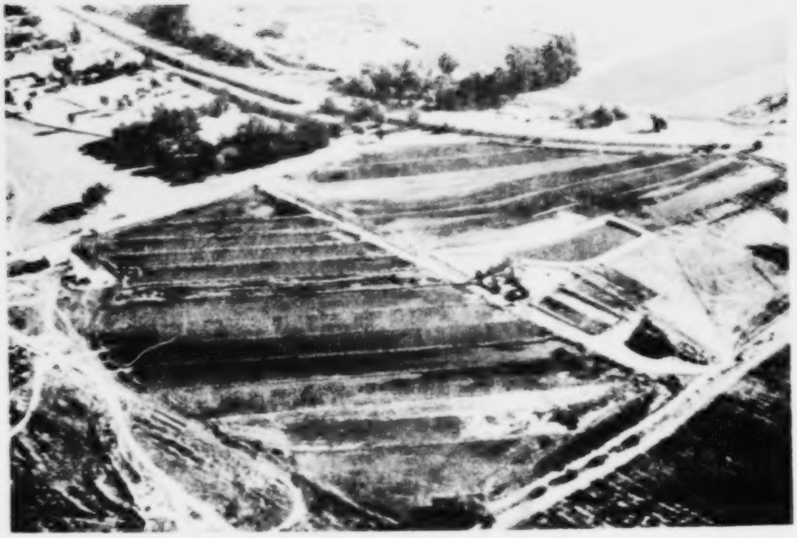

Figure 2. Facilities plan of Hollister rapid infiltration system.

The clarified effluent flows by gravity to the effluent treatment area, which consists of 20 individually controlled infiltration basins. Basins are controlled by a combination of alfalfa valves and manually prepared ditches. One additional basin is kept in reserve for overflow effluent caused by sporadic hydraulic overloading in the gravity distribution system.

The total basin area is 8.8 ha (21.7 acres) and the individual basins range from 0.3 to 0.7 ha (0.6 to 1.8 acres). Normally, wastewater is applied to a depth of 30 cm (12 in.), whereas time of application is controlled by the area of the basin. Two basins are flooded simultaneously if the daily flow exceeds the capacity of any individual basin. This situation occurs in the smaller basins (<0.37 ha, <0.9 acres) during the wet winter months.

The interval between wastewater applications ranged from 14 to 21 days during the study period. The length of time a basin was flooded and the interval between applications decreased in the cooler and wet winter months by 25 to 30%.

The initial effluent intake rate was much higher than the final intake rate, because clogging of the soil surface by effluent solids reduced the rate at which water entered the soil. The effluent was completely contained within the basin it was applied to, leaving the site only as subsurface flow. The average effluent intake rate was 10 cm/d (4 in./d) when basins are flooded to a depth of 30 cm (12 in.).

Geologically, the site is located on alluvial deposits of the San Benito River. The surface soil from which the basins are constructed is characterized as Metz sandy loam [3]. According to the Soil Conservation Service (SCS), soil permeability for Metz sandy loam ranges from 6.4 to 12.8 cm/h (2.5 to 5.0 in./h) in the upper 3 m (10 ft), and 12.8 to 25.4 cm/h (5.0 to 10.0 in./h) through the next 7.5 m (25 ft) [3].

The water table of the regional groundwater aquifer occurs at 20 m (65 ft) at the site. Eight observation wells were constructed to monitor groundwater quality at the Hollister rapid infiltration site. The depth of each well from the local ground surface is as follows:

Well	Depth, m	(ft)
1A	6	(20)
1B	24	(80)
1C	48	(160)
2A	54	(180)
3A	7.6	(25)
3B	21	(70)
4B	18.3	(61)
5A	10.5	(35)

The wells are 5 cm (2 in.) in diameter constructed from PVC pipe with a 0.6 m (2 ft) length of PVC well screen at the installed depth. Existing offsite production wells were also monitored. Well 4C (City of Hollister) serves as an upgradient control well with perforations beginning at 56 m (185 ft). Four production wells (6C, 7C, 8C, and 9C) serve as control wells downgradient of groundwater flow from the site. Observation well locations are shown in Figure 3.

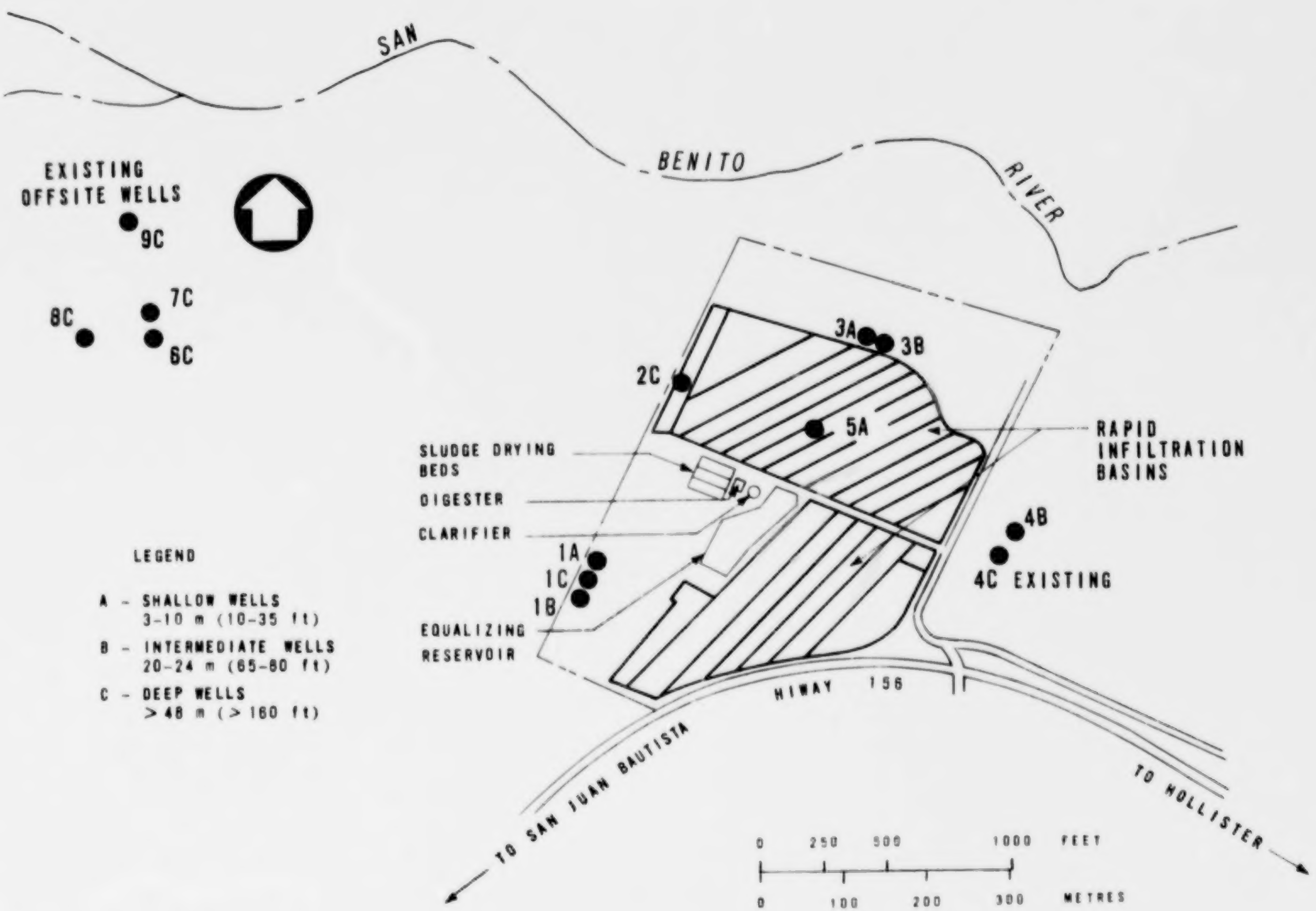
SAN
BENITO
RIVER
EXISTING
OFFSITE WELLS
9C
7C
8C
6C
3A
3B
2C
5A
RAPID
INFILTRATION
BASINS
SLUDGE DRYING
BEDS
DIGESTER
CLARIFIER
1A
1C
1B
EQUALIZING
RESERVOIR
4B
4C EXISTING
HIWAY 156
TO SAN JUAN BAUTISTA
TO HOLLISTER
LEGEND
A - SHALLOW WELLS
3-10 m (10-35 ft)
B - INTERMEDIATE WELLS
20-24 m (65-80 ft)
C - DEEP WELLS
> 48 m (> 160 ft)
0
250
500
1000
FEET
0
100
200
300
METRES

Figure 3. Observation well locations at the Hollister rapid infiltration site.

SECTION 5

RAPID INFILTRATION SYSTEM OPERATION

SYSTEM HISTORY

Wastewater was first applied to the site in 1922, when an outfall sewer was constructed to convey the untreated wastewater away from the city. At that time the site was in pasture, and the wastewater was allowed to flood the general area in a more or less uncontrolled manner. During the 1920s, 1930s, and 1940s, 10 acres of walnuts were irrigated with a portion of the wastewater until the orchards were abandoned. The wastewater could be diverted from the pasture area easily and was used mainly because it was more convenient than pumping from the native groundwater. Wastewater irrigation of nursery stock was practiced successfully at the site from 1935 to 1946. The practice was abandoned when the new infiltration basins were constructed.

In 1927, an Imhoff tank was constructed to remove settleable solids from the wastewater before it was applied to the land. In about 1946, Basins 1 through 13 (north basins) covering 4.9 ha (12.2 acres) were constructed as shown previously in Figure 2. Intermittent application of wastewater to the infiltration basins was initiated at this time. No recovery or reuse of the percolated effluent has ever been attempted.

In 1962, as a result of increased wastewater flow, a gravity clarifier was constructed to increase solids removal before applying the effluent to the infiltration basins. At that time, the original Imhoff tank was converted to an unheated sludge digester by removing the interior partitions. Seven additional basins (14 through 20) covering 4.0 ha (9.9 acres) were constructed in the late 1960s when hydraulic loading began to exceed the capacity of the original basins (see Figure 2).

In 1973, an equalizing reservoir was excavated to accept peak wastewater flows before entering the clarifier. Even with the equalization reservoir, however, hydraulic capacity of the clarifier was exceeded, resulting in high concentrations of unsettled solids being applied to the basins. Construction of the reservoir removed about 0.4 ha (1.0 acre) of infiltration basins from service; however, Basins 14 through 20 were simultaneously expanded. The total infiltration basin area including an overflow area is currently 9.1 ha (22.6 acres). The net area used for infiltration is currently 8.8 ha (21.7 acres).

A slaughterhouse and a corrugated paper recycler were the only significant industrial waste sources. The slaughterhouse waste had a high BOD while the recycle operation added a significant solids load. The total flow from both sources of 0.9 L/s (0.02 Mgal/d) and 11.0 L/s (0.25 Mgal/d), respectively, amounted to about 27% of the total flow.

CLIMATE

The climate in the Hollister area is marked by warm dry summers and cool rainy winters, which is characteristic of the Central Coastal Region of California. Although Hollister is separated from the Pacific Ocean by coastal mountains, temperatures are moderated by the ocean influence. The wet season is normally mid-November through mid-April. On the average, little or no precipitation occurs from May through August. July is the month with the highest average maximum temperature, while peak precipitation usually occurs in January. The annual evaporation exceeds precipitation in the Hollister area by several times; but the exact ratio will vary depending on annual temperature, precipitation, and wind. Mean temperatures, precipitation, and evaporation are compared to those for the 1962-1963 reporting period (only complete data available) in Table 2 [4]. The mean annual "net evaporation" is approximately 150 cm (59 in.) according to one report [5]. The net evaporation for 1962-1963 was 98 cm (39 in.) in which below average temperatures but above average precipitation occurred. The mean annual net evaporation is approximately 6% of the current annual wastewater applied to the infiltration basins.

TABLE 2. HISTORICAL CLIMATIC DATA, HOLLISTER, CALIFORNIA

	Mean			Avg temperature, °C[a]		1962-1963 water year [4][b]	
Month	Temperature, °C [6][b]	Precipitation, cm [6]	Evaporation, cm [5]	Maximum	Minimum	Precipitation, cm	Evaporation, cm
Jul	20.0	0.03	27.56	25.5	8.0	0	22.96
Aug	19.5	0.05	25.98	28.5	9.0	Trace	20.68
Sep	20.0	0.46	20.57	25.5	8.0	Trace	13.00
Oct	17.0	1.24	14.96	24.5	6.0	1.83	9.07
Nov	13.0	3.30	8.38	21.0	3.5	0.64	8.00
Dec	10.0	6.78	3.81	18.0	1.0	4.52	5.74
Jan	9.5	6.53	2.36	15.5	-1.5	9.88	4.39
Feb	11.0	6.30	4.98	20.0	8.5	7.39	6.27
Mar	13.0	4.75	9.45	18.5	3.5	5.36	8.48
Apr	14.5	2.87	16.00	18.0	5.5	6.91	7.65
May	16.5	0.89	23.62	20.5	9.0	0.99	13.06
Jun	18.5	0.20	25.91	23.5	9.0	0.25	16.71
Annual	15.0	33.40	183.59	21.5	5.5	37.77	136.02

a. Temperatures reported to nearest 0.5°C.

b. Only available evaporation data for Hollister.

cm x 0.3937 = in.

(°C x 1.8) + 32 = °F

Available climatic data for Hollister during the study period is shown in Table 3. The weather pattern encountered was highly unusual with respect to precipitation. Although the total annual rainfall was almost normal, the monthly distribution was very unusual. The largest monthly rainfall occurred in June 1976 which is normally precipitation free. Drought conditions prevailed at Hollister and throughout northern California during this time period. Based on available historical climatic data, it is estimated that the net evaporation during the study period was slightly greater than normal.

TABLE 3. CLIMATIC DATA DURING STUDY PERIOD, HOLLISTER, CALIFORNIA [7]

Month	Avg temperature, °C[a]		Hours, >38°C	Hours, <7°C	Hours, <0°C	Precipitation, cm	Relative humidity,%
	Maximum	Minimum					
May 1976	24.0	6.5	1	56	--	0.0	37.5
Jun	31.0	9.5	14	28	--	6.45	32.6
Jul	27.0	11.0	--	--	--	0.0	41.4
Aug	27.0	12.0	--	--	--	3.07	41.9
Sep	26.0	11.0	1	--	--	7.72	41.0
Oct	25.0	8.5	--	43	--	1.27	40.8
Nov	20.0	5.1	--	168	22	2.29	44.0
Dec	17.5	-1.0	--	414	83	3.15	31.5
Jan 1977	14.5	1.0	--	418	61	2.13	47.2
Feb	19.5	3.0	--	250	9	1.14	34.9
Mar	18.0	1.0	--	279	21	2.21	28.5
Apr	24.0	4.5	--	195	--	0.79	26.9
May	21.0	7.5	--	59	--	2.21	35.9
Jun	27.0	11.0	--	--	--	0.43	38.0
Jul	30.5	10.5	--	--	--	0.0	30.5
Aug	28.5	13.0	2	--	--	0.0	37.2

a. Values reported to the nearest 0.5°C.

cm x 0.3937 = in.

(°C x 1.8) + 32 = °F

SURFACE WATER HYDROLOGY

The San Benito River is the only natural surface water source that potentially has an effect on the Hollister rapid infiltration site. The San Benito River is generally dry, except during unusually heavy rainfall occurring below the point of the Hernandez Dam located some 80 km (50+ miles) upstream of the study site. Regular releases from the Hernandez Reservoir provide managed recharge of subsurface aquifers as well as flood control. Surface flow does not generally occur as a result of water release from the reservoir in the vicinity of the study site. However, groundwater levels of the main Hollister-San Juan groundwater basin in the site vicinity can be affected by the releases from Hernandez Reservoir.

WATER AND WASTEWATER CHARACTERISTICS

The mineral content of the wastewater can be traced directly to the quality of the domestic water supply. Potable water is derived solely from wells and the quality varies from source to source. In general, the water supply is relatively hard and high in total dissolved solids, sodium, chloride, sulfate, and boron. A summary of available water supply quality data for the City of Hollister is shown in Table 4. The effluent wastewater quality results, in which three 24-hour composite samples were taken each quarter for 1 year, are shown in Table 5. Data for each primary effluent composite sample taken are shown in Appendix A.

TABLE 4. CITY OF HOLLISTER WATER SUPPLY QUALITY[a]
mg/L Unless Otherwise Noted

Item	Well No.2	Well No.4	Well No.5	Cienega well
Flow, L/s	61.3	100.7	70.1	21.9
General mineral analysis				
Hardness ($CaCO_3$)	371	374	388	81
Bicarbonate ($CaCO_3$)	324	328	360	--
Carbonate ($CaCO_3$)	0	0	0	--
Hydroxide	0	0	0	--
Alkalinity ($CaCO_3$)	324	328	360	--
Ca	63	61	68	--
Mg	52	54	53	--
Fe	0.0	0.0	0.2	--
Mn	0.0	0.0	0.0	--
Na	132	130	130	16
Cl	103	101	104	20
SO_4	248	264	260	16
F	0.7	0.7	0.7	--
NO_2-N and NO_3-N	3	2	3	1.8
pH, units	7.6	7.7	7.6	--
Conductivity, μmhos/cm	1380	1380	1490	250
TDS	870	870	940	157
Trace elements				
As	0.00	0.00	0.00	--
B	0.6	0.7	0.6	0.05
Cu	<0.1	<0.1	<0.1	--
Pb	0.02	0.02	0.02	--
Se	0.00	0.00	0.00	--
Zn	0.02	0.02	0.02	--
General physical analysis				
Color, units	5	5	7	--
Odor, threshold No.	1	1	1	--
Turbidity, JTU	0.5	0.3	0.5	--

a. From City of Hollister Water Supply Monitoring data 1973.

L/s x 0.0228 = Mgal/d

TABLE 5. CITY OF HOLLISTER EFFLUENT WASTEWATER QUALITY
mg/L Unless Otherwise Noted

Constituent	Range[a]	Jul 1976[b]	Sep 1976[b]	Dec 1976[b]	Mar 1977[b]	Average
COD	546-1029	624	593	659	946	706
BOD	134-414	173	208	153	346	220
TOC	240-264	--	--	--	248	248
Total N	29.7-58.5	42.2	46.7	30.9	40.9	40.2
NH_3-N	19.7-44.0	22.2	33.5	22.3	23.2	25.3
N-organic	6.7-21.8	19.4	12.9	8.1	17.5	14.5
NO_3-N	0.16-0.8	0.7	0.34	0.5	0.18	0.43
Total P	10.0-21.5	12.7	10.7	16.2	10.2	12.4
PO_4-P	9.0-13.2	12.7	9.0	10.6	9.8	10.5
Total coliforms, count/100 mL	4.6×10^6 92×10^6-	2.2×10^6	18×10^6	69×10^6	21×10^6	27.6×10^6
Fecal coliforms, count/100 mL	3.5×10^6 24×10^6	--	6.7×10^6	11.6×10^6	19×10^6	12.4×10^6
pH, units	7.0-8.1	7.0	7.3	7.8	7.2	7.3
TDS	1016-1593	1197	1559	1033	1044	1208
Conductivity, µmhos/cm	1230-2480	1847	2400	1620	1293	1790
SS	206-438	--	--	221	327	274
Alkalinity	416-465	--	--	433	459	446
B	0.95-1.8	1.1	1.8	1.4	1.3	1.4
F	0.33-1.08	0.6	0.57	0.37	1.08	0.66
Na	196-391	246	377	230	197	262
Ca	31-71	59	71	53	31	54
Mg	27-76	62	57	75	63	64
K	10.9-16.0	11.3	12.1	12.6	15.6	12.9
Cl	185-490	258	471	192	216	284
SO_4	161-250	176	219	243	215	213
Ag	<0.005-0.012	0.007	<0.006	0.011	<0.007	<0.008
As	--	<0.01	<0.01	<0.001	<0.01	<0.01
Ba	<0.1-0.24	<0.1	<0.1	0.11	0.21	<0.13
Cd	<0.001-0.008	0.004	<0.001	0.007	0.003	<0.004
Co	<0.006-0.008	0.008	0.011	<0.006	<0.006	<0.008
Cr	<0.004-0.036	<0.004	<0.004	0.015	0.032	<0.014
Cu	0.019-0.070	0.024	0.011	0.068	0.032	0.034
Fe	0.14-0.82	0.41	0.17	0.34	0.63	0.39
Hg	--	0.001	<0.001	<0.001	<0.001	<0.001
Mn	0.055-0.094	0.069	0.064	0.068	0.080	0.070
Ni	0.015-0.092	0.063	0.086	0.015	0.040	0.051
Pb	0.012-0.12	0.016	0.033	0.068	0.10	0.054
Se	--	<0.001	<0.001	<0.001	<0.001	<0.001
Zn	0.010-0.090	0.015	0.032	0.053	0.09	0.048
SAR	4.67-8.08	5.33	8.08	4.76	4.67	5.71

a. Range of twelve 24 h composite samples.
b. Average of three 24 h composite samples.

CONSTITUENT LOADING

Wastewater characteristics that can influence the amount of wastewater that can be applied to a rapid infiltration site include organic matter, nitrogen, and phosphorus. The current constituent loading rates at the Hollister rapid infiltration site are shown in Table 6.

TABLE 6. WASTEWATER CONSTITUENT LOADING RATES
kg/ha

Constituent	Daily	Annual
COD	304	111,000
BOD	95	34,500
TOC	107	39,000
Total nitrogen	17.3	6,310
Total phosphorus	5.3	1,950

a. Based on 438 L/s applied to 8.8 ha.

kg/ha x 0.89 = lb/acre

The mechanisms which cause removal and the fate of these and other constituents during infiltration and percolation through the soil profile are discussed in the soil and groundwater investigation sections.

SECTION 6

METHODOLOGY

SITE SELECTION

The initial phase of the project involved the selection of a suitable study site. The 1974 EPA Municipal Waste Facilities Inventory for California was used to compile the initial list of candidate sites [8]. Listed in the inventory were a total of 106 sites where some variation of the rapid infiltration process was being used (see Table 7).

The candidate sites were screened according to criteria established by the EPA and additional criteria developed during the screening procedure. To be considered, a site must have been representative of the current state-of-the-art for rapid infiltration treatment as defined by the site screening criteria shown in Table 8.

The initial screening was performed using in-house information, an EPA list of existing land application systems [2], and information obtained by telephone. The most promising sites were identified for more detailed screening, and the number of candidate sites was systematically reduced. Detailed screening resulted in a list of the six most promising sites, shown in Table 9. Three promising sites that currently pretreat to the primary level and three that pretreat to the secondary level before land application were selected.

Hollister was chosen as best representing the soil and groundwater conditions for a typical site with potential for treatment by rapid infiltration where primary effluent is applied. Fontana was rejected because the depth to groundwater was very large and the direction of groundwater flow was difficult to define. Gilroy was rejected because of shallow groundwater and significant influence from a nearby industrial wastewater land application system.

Of the systems applying secondary effluent, Santee was selected as having a well-defined groundwater flow pattern. Santa Maria was rejected because of operational changes in the last few years. Oakdale was rejected because of shallow groundwater.

The final screening step included visits to the two most promising sites--Hollister and Santee. It was concluded that the most suitable site for the purposes of this study would be the one at Hollister, California. The principal reasons for selection were that Hollister was more typical of most rapid infiltration systems, had a longer history of operation, and was applying primary effluent.

TABLE 7. LIST OF CANDIDATE RAPID INFILTRATION SITES IN CALIFORNIA, OCTOBER 1975 [8]

	Community	Flowrate. L/s		Community	Flowrate, L/s
1.	Altaville	<3.1	54.	Montalvo	4.8
2.	Anderson	33	55.	Mount Shasta	22
3.	Arden	298	56.	Nevada City	31
4.	Atascadero	11	57.	Oakdale	48
5.	Avenal	20	58.	Oceanside La Saline Plant	140
6.	Big Bear Lake	8.8	59.	Oceanside Buena Vista Plant	18
7.	Brea	2.2	60.	Oceanside San Luis Rey Plant	22
8.	Buellton	3.5	61.	Olivehurst	33
9.	Buttonwillow	<3.9	62.	Orange Co. SD Plants 1 and 2	6790
10.	Calpella	<0.9	63.	Moulton Niguel Plant 2A	11
11.	Camarillo	114	64.	Orange Cove	11
12.	Capistrano Beach	17	65.	Palm Springs	131
13.	Cawelo	<5.3	66.	Perris	13
14.	Ceres	42	67.	Pinedale	11
15.	Chester	13	68.	Pixley	4.4
16.	Corona	136	69.	Sierra Lakes	2
17.	Crescentia	0.9	70.	Squaw Valley	0.4
18.	Dixon	13	71.	Porterville	75
19.	Hemet	140	72.	Rainbow A	0.35
20.	Escalon	8.8	73.	Rainbow B	0.4
21.	Etna	4.4	74.	Rainbow C	0.2
22.	Fillmore	18	75.	Rancho Santa Fe	3.2
23.	Firebaugh	13	76.	Redding (Clear Creek)	158
24.	Fontana	101	77.	Redding (Paso Land Co.)	1.8
25.	Ft. Jones	4.4	78.	Redlands	110
26.	Fresno Co. 1	8.8	79.	Reedley	35
27.	Fresno Co. 2	0.9	80.	Rialto	92
28.	Gilroy	101	81.	Richvale	1.3
29.	Gridley	39	82.	Rio-Dell	15
30.	Gustine	48	83.	Ripon	13
31.	Healdsburg	22	84.	Riverbank	13
32.	Hollister	35	85.	Salida	18
33.	Hughson	12	86.	San Clemente	88
34.	Irvine Ranch	153	87.	San Diego Co. (Brown Field)	1.3
35.	Jamestown	4.4	88.	San Miguel	1.3
36.	Jurupa	39	89.	Santa Maria	188
37.	King City	18	90.	Santa Paula	74
38.	Lake Arrowhead	22	91.	Santee	44
39.	Lakeport 1	8.8	92.	Shasta Co.	0.09
40.	Lakeport 2	4.4	93.	Soledad	13
41.	Lakeside	24	94.	Spreckels	3.5
42.	Laton	4.4	95.	Tahoe City	5.2
43.	Lodi	197	96.	Tranquillity	2.2
44.	Lone Pine	8.8	97.	Tuolumne	4.4
45.	Lechuza	<4.4	98.	Visalia	210
46.	Malibu	<4.4	99.	Waterford	8.7
47.	Trancas (L.A. Co.)	<4.4	100.	Westwood	8.7
48.	Whittier Narrows (L.A. Co.)	661	101.	Wheatland	5.7
49.	L.A. Co. SD #22 Azuza	31	102.	Winters	8.8
50.	Manteca (Raymus Village)	0.4	103.	Winton	15.0
51.	Mariposa	2	104.	Woodbridge	8.8
52.	Mendota	4.4	105.	Yreka	44
53.	Mills-Cordova	8.8	106.	Yuba City	66

L/s x 0.0228 = Mgal/d

TABLE 8. INITIAL SITE SCREENING CRITERIA

Item	Criteria
Period of operation	>10 years
Preapplication treatment	Remain unmodified for at least 10 years
Wastewater sources	Domestic and commercial[a]
Wastewater flowrate	>4.3 L/s
Wastewater application rate	>6 m/yr
Depth to groundwater	>3 m and <30 m
Control site	Comparable geohydrologic characteristics within 1.6 km of site
Availability of data	Historical wastewater and groundwater quality must be available for comparison purposes
Operation practices	1. Wastewater application to the spreading basins must be intermittent[b] 2. Sludge must never have been applied to the spreading basins[c] 3. Soil conditions in the basins should not have been altered drastically[d]

[a] Industrial wastewater, in small amounts, resembling municipal wastewater is acceptable.

[b] Systematic flooding and drying over several days with multiple independent basins.

[c] Constituents are generally much more concentrated in sludge than in wastewater.

[d] Surface disking or scarifying to restore infiltration is normal and acceptable.

L/s x 0.0228 = Mgal/d
m x 3.281 = ft
km x 0.621 = mi

TABLE 9. SUMMARY OF DESIGN AND OPERATING INFORMATION ON SIX CANDIDATE SITES

Site location	Level of pretreatment	Flowrate, L/s	Application rate, m/yr[a]	Duration of operations, total number of years
Hollister	Primary	44	15	30
Fontana	Primary	110	21	20
Gilroy	Primary	118	24	52
Santee	Secondary	44	58	17
Santa Maria	Secondary	241	46	14
Oakdale	Secondary	61	67	10

[a] Average annual applied wastewater volume divided by the infiltration basin area.

L/s x 0.0228 = Mgal/d
m/yr x 3.281 = ft/yr

STUDY PLAN

The method of approach was to characterize the applied effluent, groundwater, and soil over a 1 year period. The concentration of constituents in the effluent and groundwater were compared to determine the current efficiency of the soil matrix in removing constituents contained in the effluent. The constituents of concern in groundwater were those for which drinking water standards have been set, and those that affect water quality for irrigation and livestock watering. Observation wells constructed for sampling groundwater were shown in Figure 3.

A mass balance of selected soil constituents was performed to determine the effects of 30 years of effluent application to the land on the ability of the soil to retain constituents in the applied effluent. For soils, the important constituents were those that could restrict the use of the site to grow crops if the current land use were changed. The soil sampling locations are shown in Figure 4.

A sampling program was devised to quantify the constituents in the effluent, groundwater, and soils. Certain constituents, specifically organic pesticides and viruses, were not measured because of the inordinant expense involved in the analyses. A summary of the sampling program is given in Table 10.

TABLE 10. SUMMARY OF SAMPLING PROGRAM

	Type of samples		
Feature	Effluent	Groundwater	Soil
Total No. of samples	12	52	20
No. of constituents per sample	38	37	25
Timing of samples	6/76, 9/76, 12/76, 3/77	6/76, 9/76, 12/76, 3/77	6/76, 3/77
No. of sampling locations	1	13	5
Type of samples	24 h composite	Grab	Multiple composite

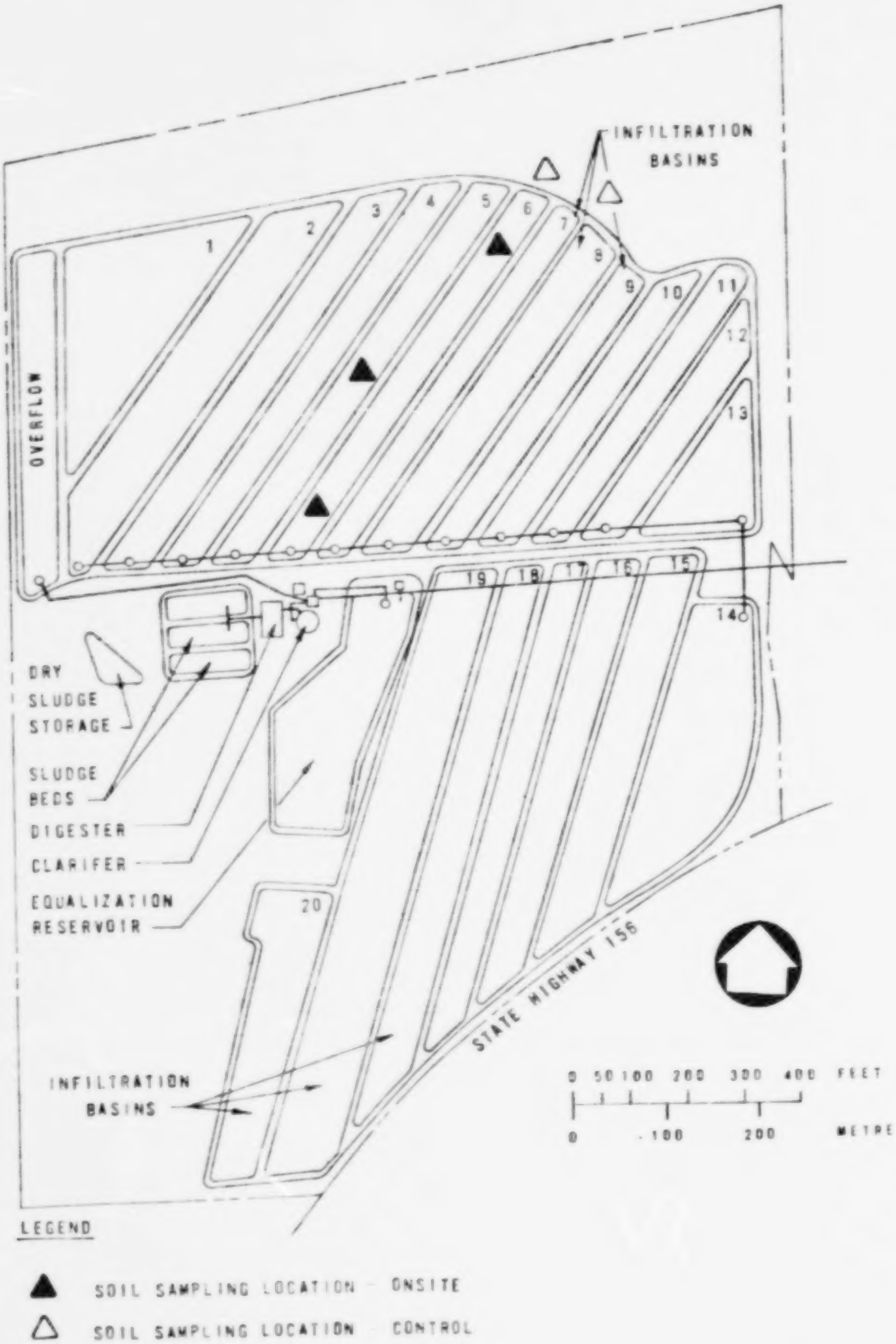
INFILTRATION BASINS
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
OVERFLOW
DRY SLUDGE STORAGE
SLUDGE BEDS
DIGESTER
CLARIFER
EQUALIZATION RESERVOIR
INFILTRATION BASINS
STATE HIGHWAY 156
0 50 100 200 300 400 FEET
0 100 200 METRE
LEGEND
▲ SOIL SAMPLING LOCATION – ONSITE
△ SOIL SAMPLING LOCATION – CONTROL

Figure 4. Proposed soil sampling locations at the Hollister rapid infiltration site.

Effluent Sampling

The primary effluent was sampled through the use of a 24 hour, refrigerated, composite sampler. The sampler was battery-powered and installed to take samples from the effluent trough of the clarifier. Each sampling day (3 consecutive days) produced 24 separate 1 hour samples that were composited according to flow. The constituents analyzed for were:

Organic: COD, BOD, TOC

Bacterials: Total coliforms, fecal coliforms

Minerals: Suspended solids, TDS, pH, alkalinity, conductance, N organic, NO_3-N, NH_3-N, total N, total P, PO_4-P, Na, K, Ca, Mg, Cl, SO_4, B, F

Trace elements: Ag, As, Ba, Cd, Co, Cr, Cu, Fe, Hg, Mn, Ni, Pb, Se, Zn

Groundwater Sampling

Sampling of the groundwater took place in the specially constructed series of observation wells and five existing offsite wells. Observation wells at Sites 1, 2, and 3 were drilled to intercept the subsurface flow leaving the site. Observation well No. 4 was drilled as an offsite control well. Observation well No. 5 was drilled near the center of the site to intercept the top of the groundwater mound created by the infiltrating effluent.

At the peripheral well locations (Nos. 1 through 4), the first drilling was to approximately 54 m (180 ft). At 1.5 m (5 ft) intervals the drilling was stopped and a splitspoon sample of material was taken. This allowed the profile to be plotted by the geologist. A well screen was established if water bearing material existed near the 54 m (180 ft) level. A second well was drilled to either an intermediate depth (24 m = 80 ft) or a shallow depth (8 to 11 m = 25 to 35 ft) or both.

It was originally estimated that four shallow wells (noted as "A"), three intermediate wells (noted as "B"), and four deep wells (noted as "C") would be drilled. The location of potential water bearing material in the soil profile was determined at each site from a log of each boring. Observation wells were established where potential water bearing material was found. The locations of the shallow, intermediate, and deep wells established from the geologic investigation were shown in Figure 3.

The wells were equipped with pumps to obtain groundwater samples from specified depths. Sampling pumps were hand crank portable plunger pumps with a flush cap cylinder, rod and couplings, and PVC carrier pipe. The flush cap cylinder was smaller in diameter than the 5.1 cm (2 in.) inside diameter of the observation wells. The pumps were able to obtain sufficient water volume to flush out the standing water column in a relatively short period of time. Samples were withdrawn in June 1976, September 1976, December 1976, and March 1977 (see Table 10).

Five existing offsite wells were monitored at the same frequency. An upstream well (No. 4C) owned by the City of Hollister served as a supplementary control well. Four downstream wells (6C, 7C, 8C, 9C) 0.8 km (0.5 mile) away were also utilized. These privately owned wells are used for irrigation and domestic water production.

Suspended solids content was measured selectively in the groundwater and carbon chloroform extract (CCE) was not measured in the primary effluent. All other constituents measured in the groundwater were the same as for the applied effluent. The methods of analyses for both effluent and groundwater samples were from Standard Methods [9].

Soil Sampling and Analytical Procedures

The soil profile was sampled at three onsite locations and two offsite control locations (Figure 4). The onsite locations were in a spreading basin that has been in use since about 1946. Onsite and control samples were taken in June 1976, and again in March 1977. To establish the extent of vertical variation in soil chemistry within the soil profile, samples were taken at the following depth increments:

Depth increment No.	cm	(ft)
1	0-2	0-0.07
2	2-4	0.07-0.13
3	4-16	0.13-0.52
4	25-35	0.8-1.2
5	95-105	3.1-3.4
6	295-305	9.7-10.0

Composite samples were taken at each sampling location to account for the spatial variability of the soil. The basin selected for sampling was 18 by 180 m (60 by 600 ft); accordingly, grid nodes spaced 1.5 by 3 m (5 by 10 ft) were established at each sampling location. For the first three depth increments, 20 subsamples were taken for compositing (one per grid node). Preliminary analyses indicated that increments 1 through 3 were indistinguishable and were finally analyzed as one composite whose mass was proportional to its volume. For the fourth depth increment at the onsite locations, 10 subsamples were composited. The fifth and sixth increments at the onsite locations and the last three depth increments at the control site were sampled from trenches excavated by backhoe. Three equally spaced trenches were excavated over the original grid nodes. Samples were taken from each face of a trench to duplicate as closely as possible the 10 subsample locations of the fourth depth increment. The high soil density at the control site made it necessary to take subsamples from the excavated trench sidewalls rather than sampling from the surface for the last three increments.

To determine if any significant quality changes have resulted from the application of wastewater, 23 chemical and 3 physical soil parameters were monitored at both control and treatment site locations. All laboratory analyses were performed on air-dried soils. A listing of the constituents and analytical procedures is shown in Table 11. The only field performed measurements were bulk density and infiltration rate.

TABLE 11. SOIL CONSTITUENTS AND ANALYTICAL PROCEDURES

Procedure	Description	Soil fraction
Laboratory		
pH [11]	1:1 paste	All
Conductivity [11]	Saturation extract	All
Organic Matter [12]	$K_2Cr_2O_7-H_2SO_4$	<0.6 mm
$CaCO_3$ [12]	Gravimetric HCl loss	<2 mm
Nitrogen [11]	Kjeldahl	<2 mm
B [11]	Hot water extract	<2 mm
Total-P [11]	$HClO_4$-digest	<2 mm
Extractable-P [11]	0.5 m $NaHCO_3$	<2 mm
CEC [11]	NH_4OAc - pH 7	<2 mm
Exchangeable cations [11]	NH_4OAc - pH 7	<2 mm
Metals [13]	DTPA-TEA extraction[a]	<2 mm
Particle Size [11]	Sieve-hydrometer[b]	All
Field		
Infiltration Rate [11]	Double cylinder	--
Bulk Density [11]	Core, excavation	--

a. 0.005 M diethylenetriamine pentacetic acid - 0.1 M triethanolamine - 0.01 m $CaCl_2$, 40 mL/20 g soil.

b. Clay analyses performed at discretion of laboratory technician.

mm x 0.03937 = in.
m x 3.281 = ft.

Additionally, phosphorus sorption experiments were conducted in the laboratory on both control and treatment site samples. Sorption isotherms were determined using a "non-destructive" method developed by Enfield and Bledsoe [10].

SECTION 7

HYDROGEOLOGIC INVESTIGATION

The rapid infiltration site at Hollister is on the flood plain of the San Benito River, approximately 150 m (500 ft) south of the river channel. The flood plain is typically flat and has been used as a source for commercial sand and gravel in the Hollister area. The site is protected from flooding by fill materials.

REGIONAL HYDROGEOLOGY

The site is located in the San Juan subbasin of the Gilroy-Hollister groundwater basin. The San Juan subbasin is defined within the larger basin by geologic features that form relatively impermeable barriers to groundwater movement. These boundaries include the Sargent anticline in the Lomerias Muertos and Flint Hills on the north, the Calaveras fault and the Bird Creek Hills on the east, and the San Andreas fault zone on the southwest (Figure 5).

The San Juan subbasin is further divided into subunits. The boundaries of these subunits are minor faults or zones of deformation that create hydraulic discontinuities. The location of the Hollister site relative to nearby subbasin and subunit boundaries is illustrated in Figure 6.

The major aquifer in the San Juan subbasin consists of unconsolidated to weakly-cemented alluvium of Holocene Age and the upper member of the Purisima formation of Pliocene age. These units are indistinguishable on drillers' or electric logs and are generally believed to form a single hydrologic unit. Within these sediments, the most productive water-bearing units are lenticular beds of sand and gravel that are interbedded with silt and clay layers. Groundwater occurs mainly under artesian or semi-artesian conditions in these units, although locally perched or semi-perched conditions may exist.

The Holocene alluvium is broken by currently active faults in places, such as the one that occurs approximately 600 m (2,000 ft) west of the Hollister site. The Purisima formation and the older Tertiary and Jurassic rocks that underlie the alluvium are highly folded and have been extensively broken and tilted by faulting. Electric logs of gas production wells in the Lomerias Muertos indicate that poor quality groundwater occurs to depths of 200 to 300 m (600 to 1,000 ft) in the older formation.

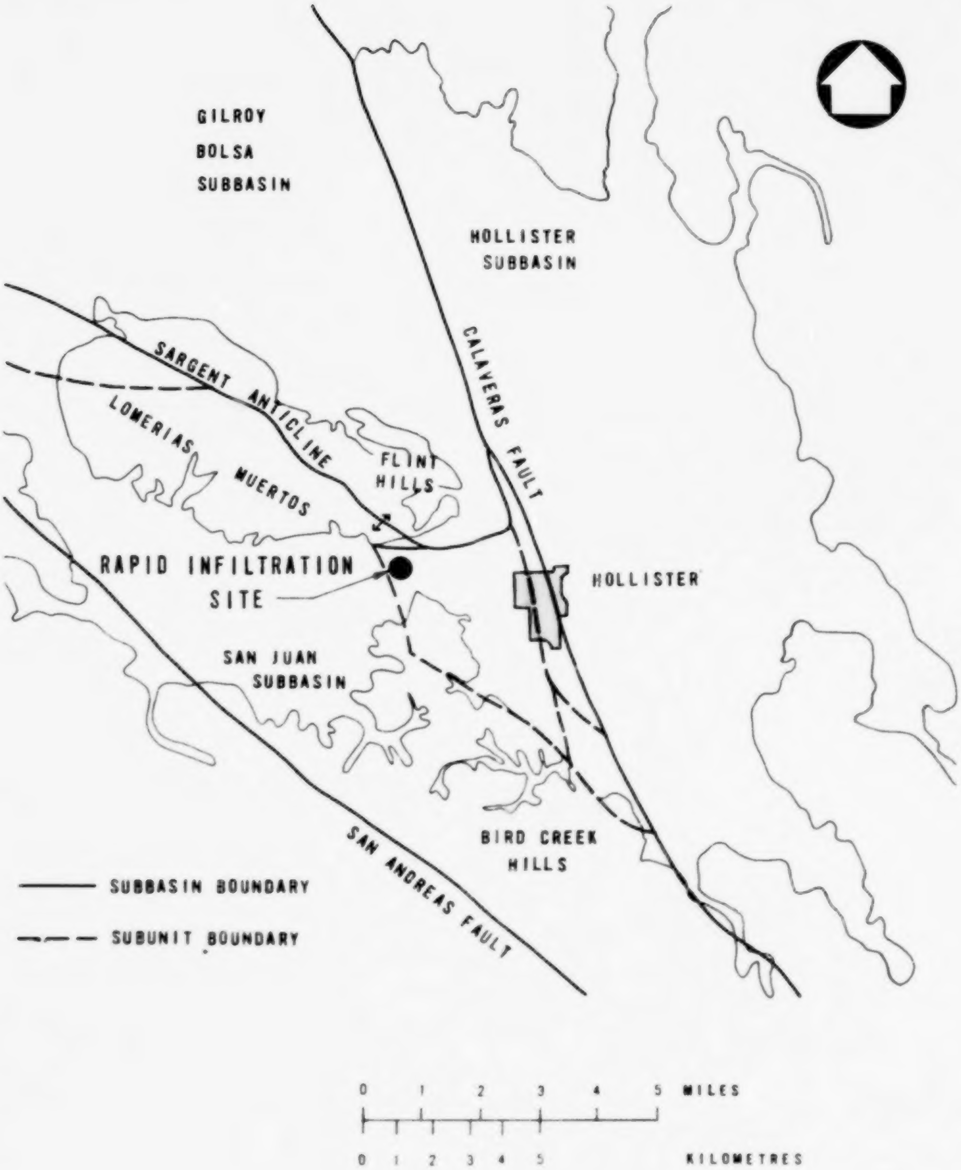
GILROY
BOLSA
SUBBASIN
HOLLISTER
SUBBASIN
CALAVERAS FAULT
SARGENT ANTICLINE
LOMERIAS MUERTOS
FLINT HILLS
RAPID INFILTRATION SITE
HOLLISTER
SAN JUAN SUBBASIN
BIRD CREEK HILLS
SAN ANDREAS FAULT
SUBBASIN BOUNDARY
SUBUNIT BOUNDARY
0 1 2 3 4 5 MILES
0 1 2 3 4 5 KILOMETRES

Figure 5. Relation of groundwater subbasins and subunits to geologic features [14].

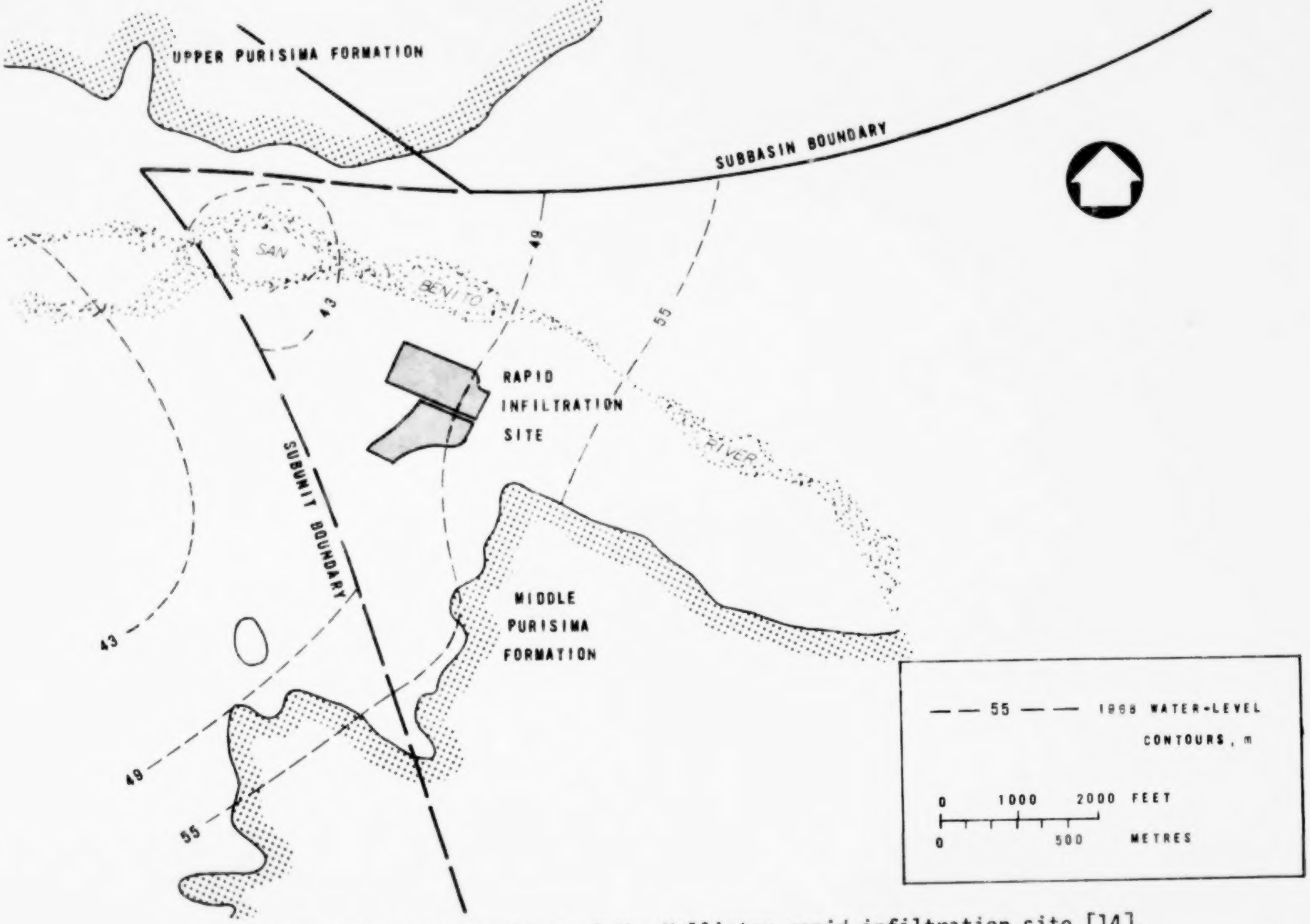
UPPER PURISIMA FORMATION
SUBBASIN BOUNDARY
SAN
BENITO
RIVER
43
49
55
RAPID
INFILTRATION
SITE
SUBUNIT BOUNDARY
MIDDLE
PURISIMA
FORMATION
43
49
55
55 1968 WATER-LEVEL CONTOURS, m
0 1000 2000 FEET
0 500 METRES

Figure 6. Hydrogeology of the Hollister rapid infiltration site [14].

Within the main groundwater body of the San Juan subbasin, groundwater flow is toward a major pumping depression in the south-central part of the valley. This depression has been caused by the withdrawal of groundwater for irrigation in amounts that are in excess of the rate of recharge, particularly since 1945. Throughout the valley, groundwater levels declined up to 30 m (100 ft) between 1913-1968, and the San Benito River changed from a gaining to a losing stream. Currently, all discharge from the San Juan subbasin is by wells, and recharge consists of stream infiltration and direct infiltration of precipitation. During the period 1946-1969, discharge exceeded recharge by an average rate of 123 L/s (2.9 Mgal/d) [15]. However, even with current drought conditions, which began in 1975, groundwater levels in the main aquifer did not decrease during the study period, beneath the rapid infiltration site.

OBSERVATION WELLS

Existing information concerning the subsurface hydrology near the site was reviewed and used as a basis for locating the groundwater monitoring wells. This information included a U.S. Geological (USGS) survey report on the regional subsurface hydrology [14], drillers' logs for wells near the site, and the logs of two shallow test borings completed at the site [16].

Examination of these data sources indicated that the principal aquifer in the vicinity of the Hollister site is alluvium and the upper member of the Purisima formation. The principal aquifer is underlain by the middle member of the Purisima formation, a less permeable unit that contains saline water at depth and water with a high concentration of sulfate locally. The base of the fresh water lies between 100 to 200 m (300 to 600 ft) below the surface in the Hollister area.

The water level contours shown in Figure 6 indicate that groundwater beneath the site moves in a westerly direction toward the subunit boundary west of the site. These water level contours are based on wells screened at depths between 30 to 100 m (100 to 300 ft).

Four peripheral sites and one site within the effluent treatment area were chosen to monitor groundwater levels and quality. Monitoring sites were located close to the infiltration basins so that extraneous effects on groundwater quality, such as former or existing upgradient waste loadings, irrigation, streamflow infiltration, etc., could be eliminated.

Observation well sites 1, 2, and 3 (see Figure 7) were located on the downgradient side of the site to intercept the suspected path of groundwater flow from the infiltration basins. It was proposed that a shallow, an intermediate, and a deep observation well be constructed at each of these three sites. The shallow wells would have a 0.6 m (2 ft) length of screen set at a depth between 3 to 13 m (10 to 40 ft), and the intermediate and deep wells would have similar screens set between 20 to 30 m (60 to 90 ft) and 45 to 60 m (135 to 180 ft), respectively. Available information indicated that the movement of percolates through the unsaturated zone might be complex and that perching conditions might be caused by apparently extensive clay deposits which underlie the site at depths of less than 10 m (30 ft). The shallow

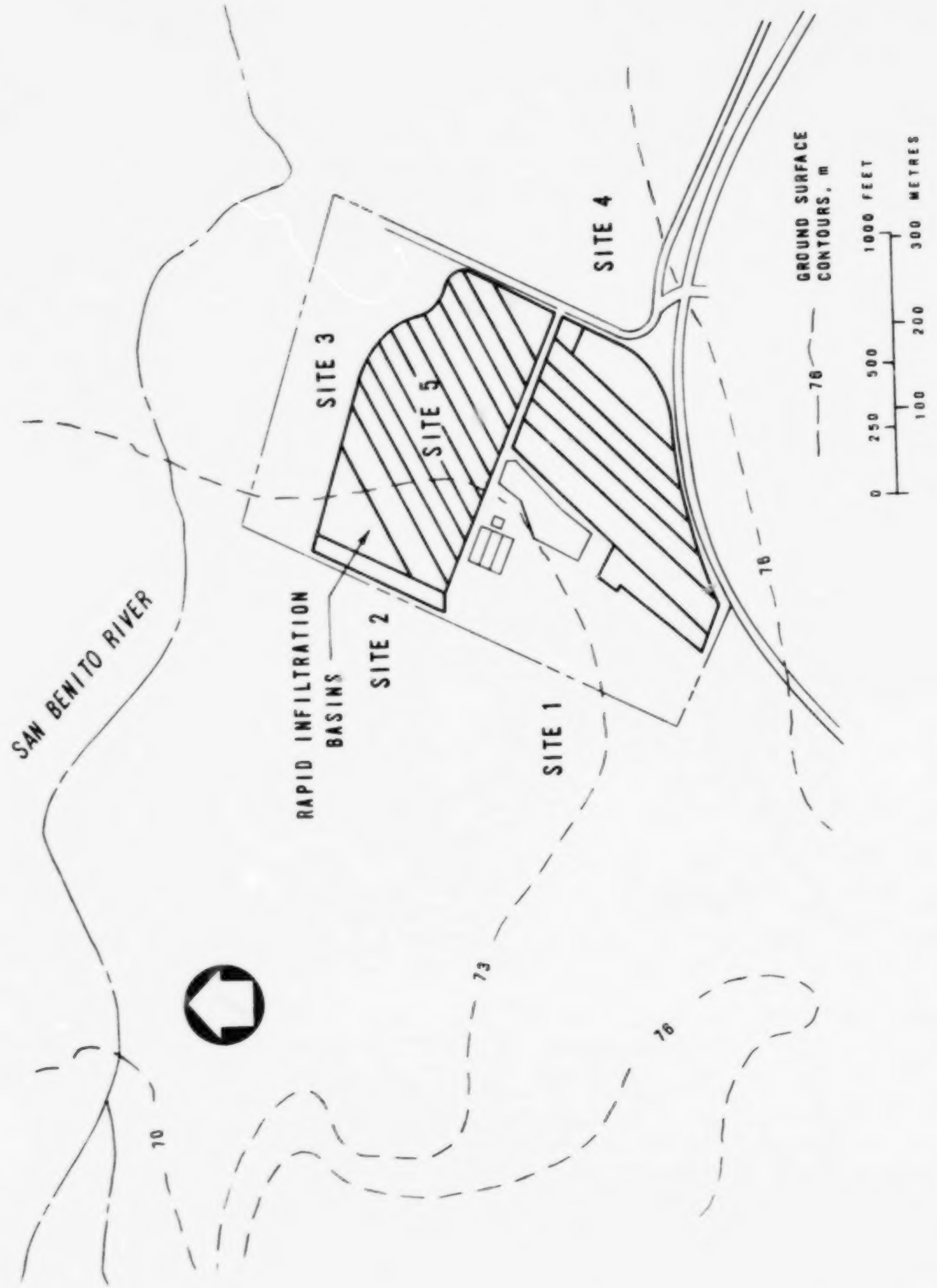
SAN BENITO RIVER
RAPID INFILTRATION BASINS
SITE 1
SITE 2
SITE 3
SITE 4
SITE 5
70
73
76
76
— 76 — GROUND SURFACE CONTOURS, m
0
250
500
1000 FEET
100
200
300 METRES

Figure 7. Observation well sites.

wells at each site were proposed to reveal possible perching in the unsaturated zone, while the intermediate and deep wells were intended to define the movement of the effluent in the saturated zone. The number of wells that were actually placed at each site was limited by the availability of permeable materials within the desired depth intervals.

At Site 4, the City of Hollister supply well was proposed as a deep monitoring well upgradient from the treatment site. Therefore, only intermediate and shallow wells were proposed at this site.

Site 5, located in infiltration basin 6, was proposed as a shallow observation well that would intercept the groundwater mound created by infiltration of the applied effluent.

The construction of the observation wells at Sites 1, 2, and 3 proceeded as follows:

1. A 12.7 cm (5 in.) diameter uncased boring was drilled to the maximum scheduled completion depth using a Failing 750 rotary drilling rig. The completion depth depended on the depth at which permeable materials were encountered in the 45 to 60 m (135 to 180 ft) interval.

2. During the drilling, 3.5 cm (1.4 in.) diameter splitspoon samples were taken at 1.5 m (5 ft) intervals throughout the boring. The location of permeable zones was observed by a geologist as the samples were taken.

3. Upon completion of the boring, depths for observation wells were selected based on availability of permeable zones. If feasible, a deep observation well was installed in the borehole. The observation wells were constructed with lengths of 5 cm (2 in.) diameter PVC pipe with a 0.6 m (2 ft) length of 10-slot PVC screen (0.03 cm, 0.01 in., openings) at the bottom. The space between the well and the open hole was filled with sand at the screened interval and with bentonite slurry from the top of the screen to the surface.

4. Similar intermediate and shallow boreholes were then drilled to the chosen depths for the installation. These wells were within 1.5 m (5 ft) of the original boring, and splitspoon samples were therefore not taken. A drawing of a typical completed observation well and a photograph of a drilling rig are shown in Figure 8.

At Sites 4 and 5, borings were drilled only as far as the intermediate and shallow zones, respectively. Similar observation wells were then installed in the boreholes.

Pertinent information concerning observation well construction is contained in Table 12. Shallow, intermediate, and deep wells are indicated by the letters A, B, and C, respectively. Examination of the table shows that, of the original 12 proposed observation wells, 4 could not be completed due to the extensive clay layers beneath the site.

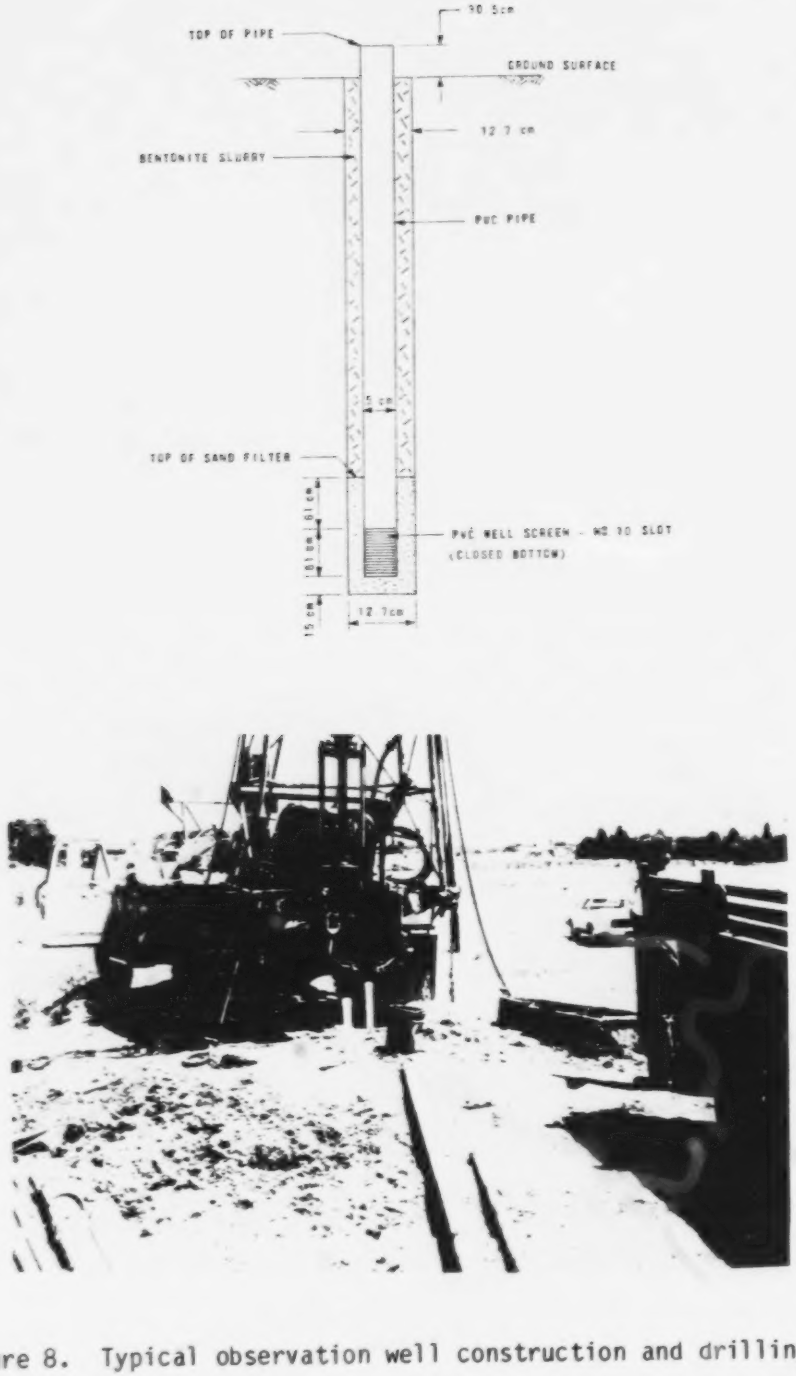
30.5cm
TOP OF PIPE
GROUND SURFACE
12.7 cm
BENTONITE SLURRY
PVC PIPE
5 cm
TOP OF SAND FILTER
61 cm
61 cm
PVC WELL SCREEN - NO 10 SLOT
(CLOSED BOTTOM)
15 cm
12.7cm

Figure 8. Typical observation well construction and drilling rig.

TABLE 12. OBSERVATION WELL CONSTRUCTION DATA

Well No.	Screened interval, m below land surface	Elevation of land surface, m	Elevation of measuring point (top of casing), m
1A	5.5-6.1	73.7	74.2
1B	23.8-24.4	73.7	74.2
1C	48.2-48.8	73.7	74.2
2C	54.3-54.9	72.1	72.7
3A	7.0-7.7	73.0	73.4
3B	20.7-21.3	73.0	73.5
4B	17.7-18.3	73.8	74.6
4C[a]	56.4-59.4	--	74.4
5A	10.1-10.7	72.8	73.4

a. Existing City of Hollister well.

m x 3.281 = ft.

SITE HYDROGEOLOGY

Graphic logs of the borings at Sites 1 through 5, which are north-south and east-west geologic cross-sections through the site area, are shown in Figures 9 and 10. These graphic logs were based on the geologist's description of the samples. The logs for two test borings that were done in a previous investigation and a driller's log for the City of Hollister supply well (observation well 4C) are also shown. Although it is evident that the nature of subsurface materials was highly variable both horizontally and vertically, several generalizations can be made.

A layer of permeable material generally occurs from the surface to a depth of 10 to 20 m (30 to 60 ft). A clay layer separates this upper permeable zone from an intermediate one which occurs at depths between 15 to 25 m (45 to 75 ft). All of the deep logs showed a thick zone of silt and clay beneath the intermediate permeable zone. A deep zone of permeable material also occurs below depths of 55 to 65 m (165 to 195 ft).

The depth to water in the observation wells that were constructed at the site varied from 10 to 20 m (30 to 60 ft) below the surface. The water level elevations, on 4 selected days, in the six observation wells that were screened in the saturated zone are shown in Figure 11. At each individual site, the water level in the intermediate zone was higher than that in the deep zone, indicating a net downward movement of water. This movement of water may be in response to the pumping from the city well (4C) in the deep aquifer.

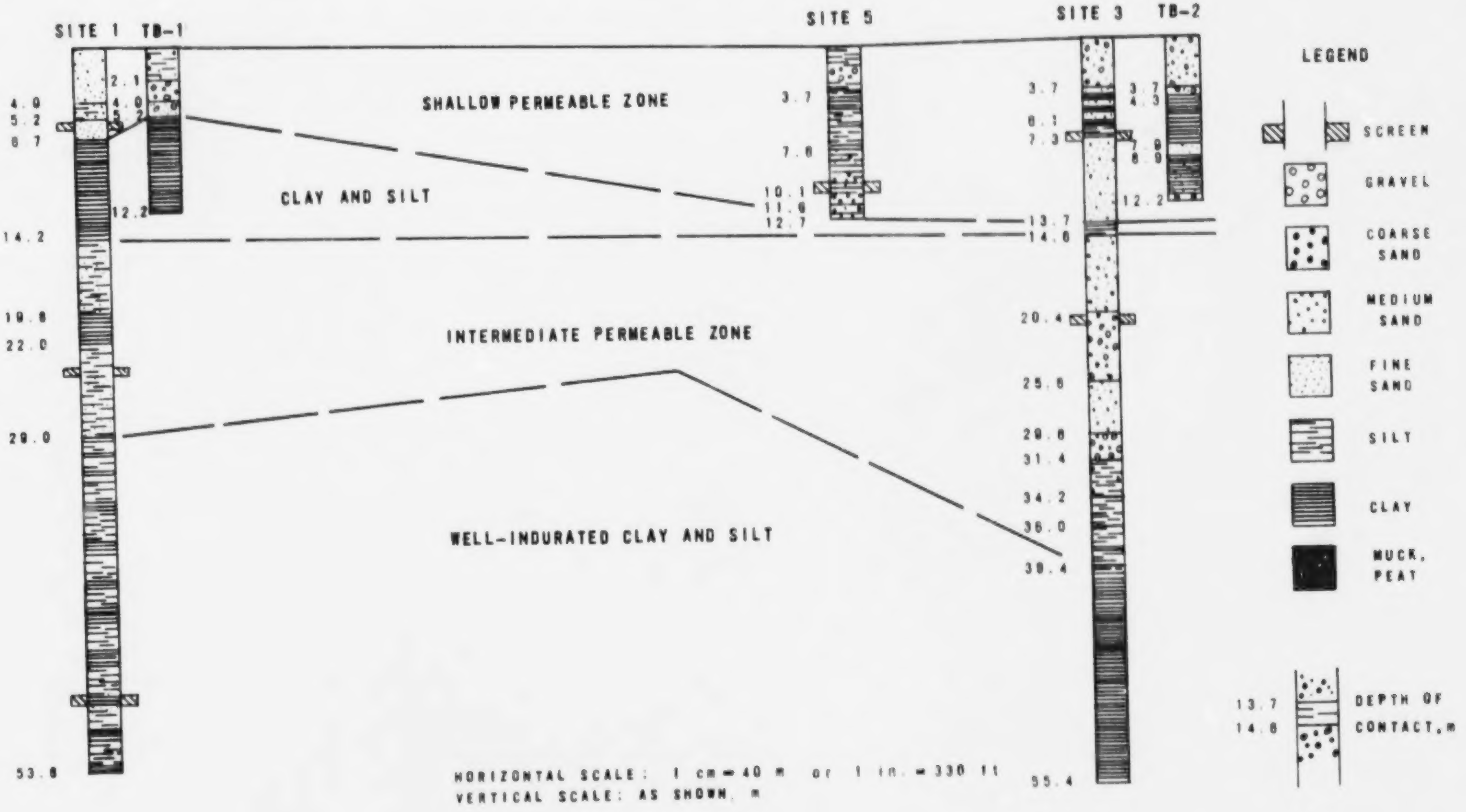
SITE 1
TB-1
SITE 5
SITE 3
TB-2
4.0
5.2
6.7
2.1
4.0
5.2
12.2
14.2
19.8
22.0
29.0
53.8
SHALLOW PERMEABLE ZONE
CLAY AND SILT
INTERMEDIATE PERMEABLE ZONE
WELL-INDURATED CLAY AND SILT
3.7
7.6
10.1
11.6
12.7
3.7
6.1
7.3
13.7
14.8
20.4
25.8
29.8
31.4
34.2
36.0
39.4
55.4
3.7
4.3
7.9
8.9
12.2
LEGEND
SCREEN
GRAVEL
COARSE SAND
MEDIUM SAND
FINE SAND
SILT
CLAY
MUCK, PEAT
13.7
14.8
DEPTH OF CONTACT, m
HORIZONTAL SCALE: 1 cm = 40 m or 1 in. = 330 ft
VERTICAL SCALE: AS SHOWN, m

Figure 9. North-south geologic section.

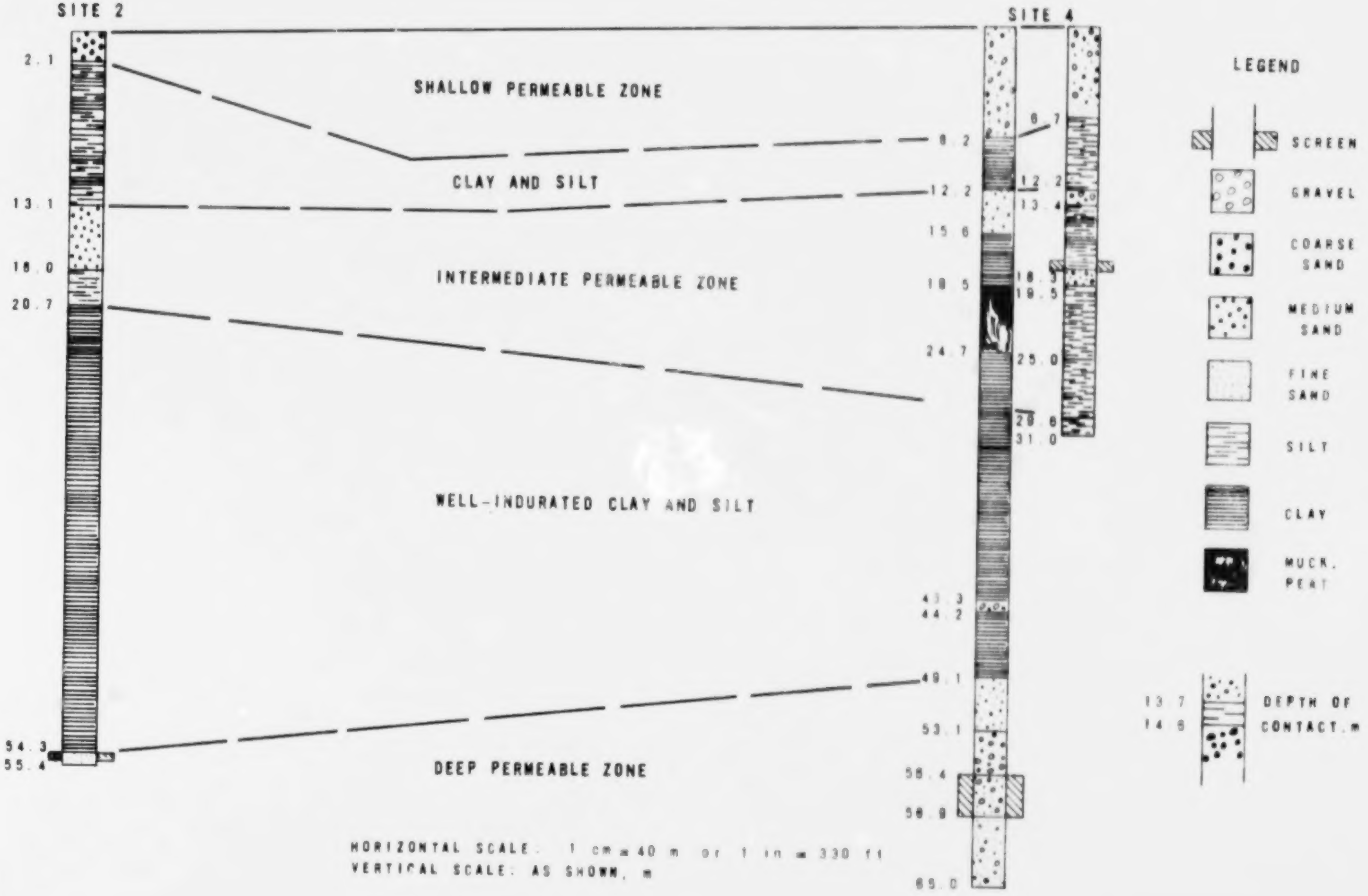
SITE 2
SITE 4
SHALLOW PERMEABLE ZONE
CLAY AND SILT
INTERMEDIATE PERMEABLE ZONE
WELL-INDURATED CLAY AND SILT
DEEP PERMEABLE ZONE
2.1
13.1
18.0
20.7
54.3
55.4
8.2
12.2
15.6
18.5
24.7
43.3
44.2
49.1
53.1
58.4
58.8
65.0
8.7
12.2
13.4
18.3
19.5
25.0
28.6
31.0
LEGEND
SCREEN
GRAVEL
COARSE SAND
MEDIUM SAND
FINE SAND
SILT
CLAY
MUCK, PEAT
13.7
14.6
DEPTH OF CONTACT, m
HORIZONTAL SCALE: 1 cm = 40 m or 1 in = 330 ft
VERTICAL SCALE: AS SHOWN, m

Figure 10. East-west geologic section.

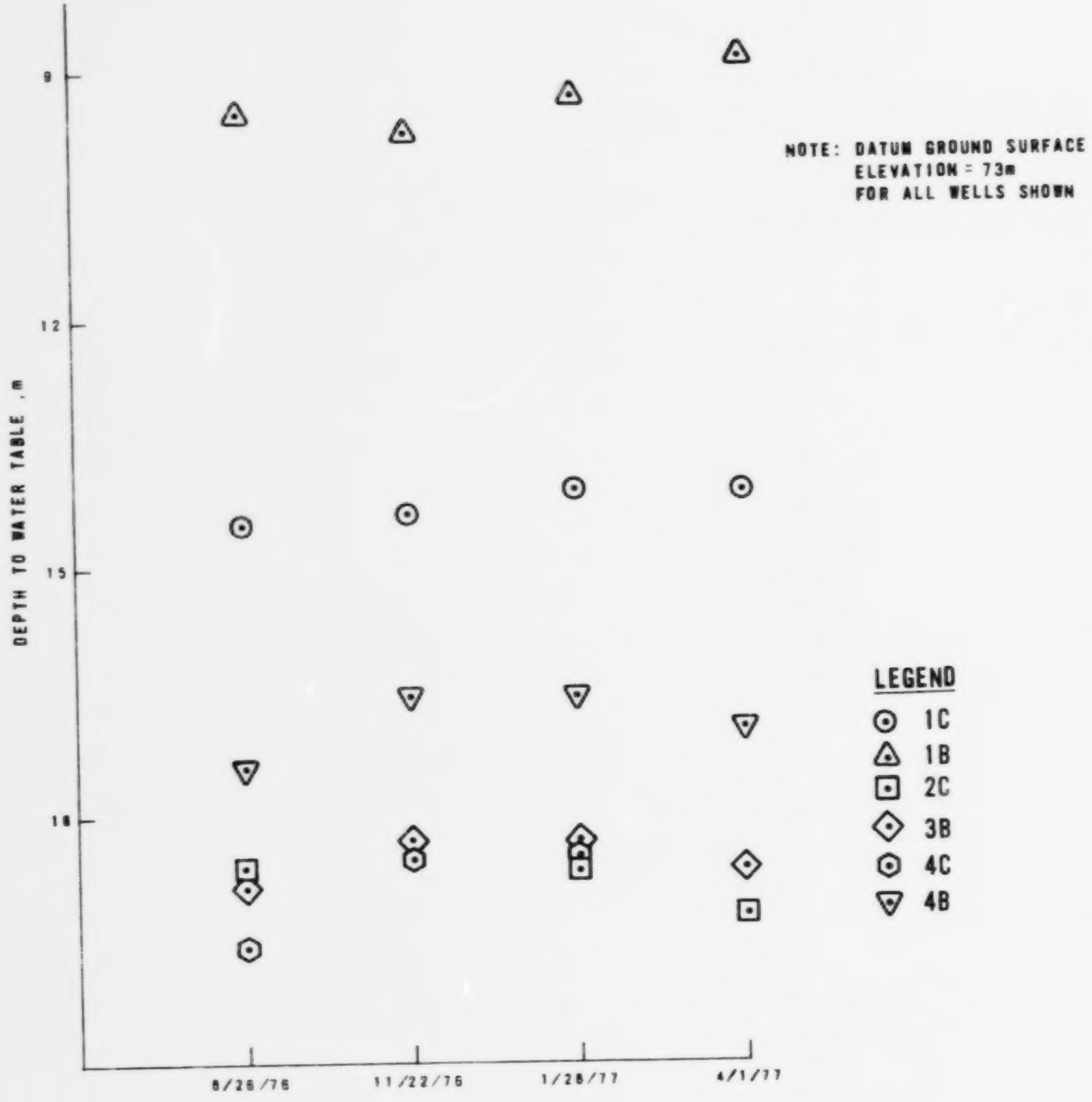

Figure 11. Water levels in intermediate and deep observation wells on four selected days.

Within the intermediate zone, the water levels indicated a movement of water toward the northeast. This trend may have been caused by greater leakage from the intermediate to the deep aquifer in the vicinity of Site 3, since the impermeable clay between the two zones appeared to be thinner to the north.

WATER LEVEL RESPONSE

Shallow Wells

The shallow observation wells at Sites 1, 3, and 5 were installed to determine if perching conditions occurred as effluent percolated down to the clay layers that underlie the upper permeable zone, as well as for monitoring water quality. Early in the 1 year study period it was discovered that Wells 3A and 5A had rapid water level responses to effluent application. Continuous water level recorders were installed to more accurately define these responses (Figure 12). Well 1A remained dry throughout the study, except when the berm around Basin 20 was breached and the field adjacent to Site 1 was flooded.

The response of Wells 3A and 5A to wastewater application is shown in Figure 13. The water level in Well 3A rose at a greater rate and to a higher elevation than that in Well 5A. This difference was most likely a result of higher permeabilities in the upper layer in the vicinity of Site 3 which created a preferred path for infiltration. After the peak water level was reached, the perched water table dropped at a greater rate in Well 3A than in Well 5A, again indicating a higher aquifer permeability at Site 3.

Both of these wells responded noticeably to flooding in Basins 3 through 8 since these basins were closest to the two wells. The effects of flooding in other basins were not evident on the hydrographs of Wells 5A or 3A.

It was also observed, during the study, that the rise of water levels in Wells 3A and 5A was related to the depth of the effluent applied in the infiltration basins. A specific relationship between these phenomena was beyond the scope of the project, however.

Intermediate and Deep Wells

The intermediate depth observation wells constructed at Sites 3 and 4 showed no discernable response to effluent application. The wells did show a yearly water level fluctuation of 0.7 to 1.0 m (2 to 3 ft), with high water levels occurring during the winter when natural recharge is greatest (Figure 14).

The intermediate well at Site 1 showed a 1 m (3 ft) rise in water level at the same time that the water level in Well 1A rose. This occurred when the adjacent field was flooded following a breach in the Basin 20 berm. The boring log from Site 1 showed 15 m (50 ft) of clay and silt between the upper and intermediate permeable zones. The most probable explanation for the response of Well 1B is that there were zones of permeable material nearby that allowed the effluent to pass down to the intermediate zone.

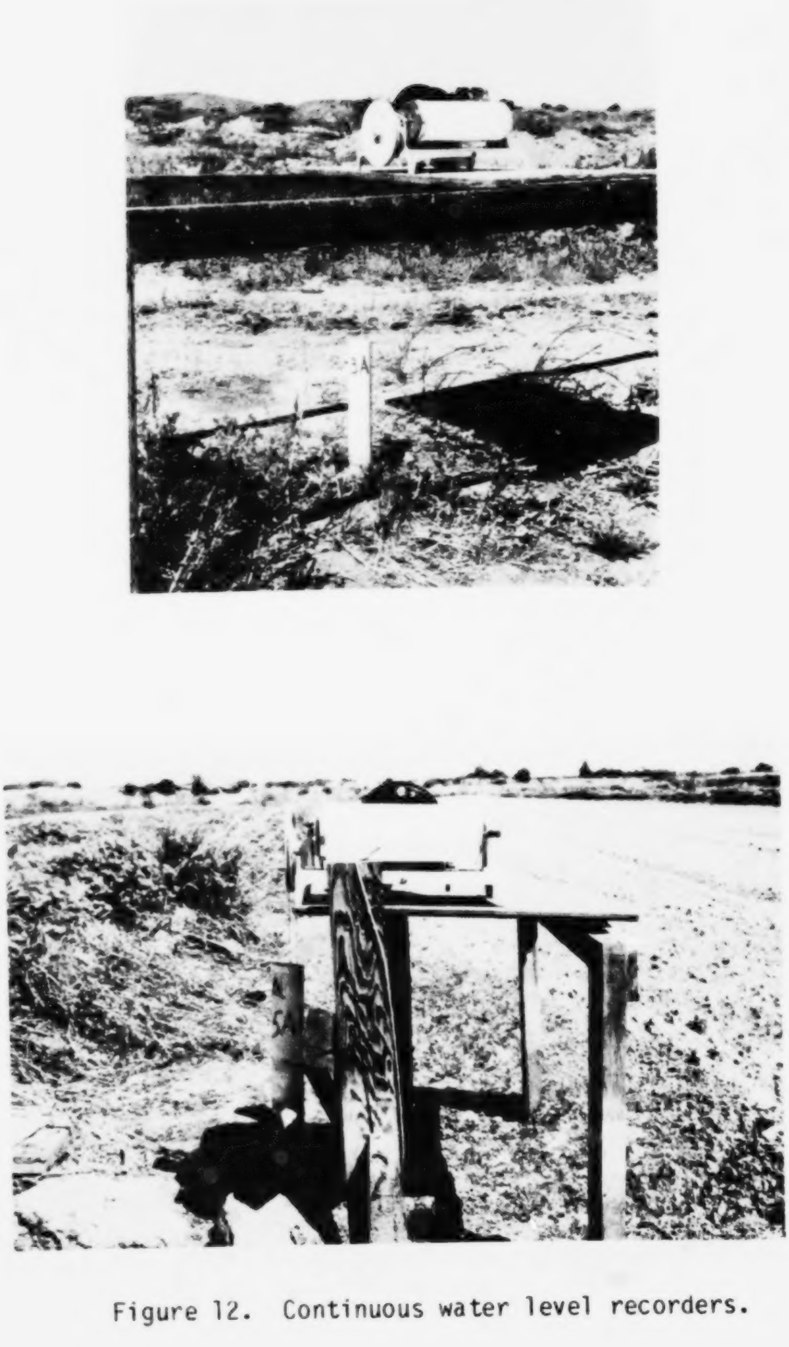

Figure 12. Continuous water level recorders.

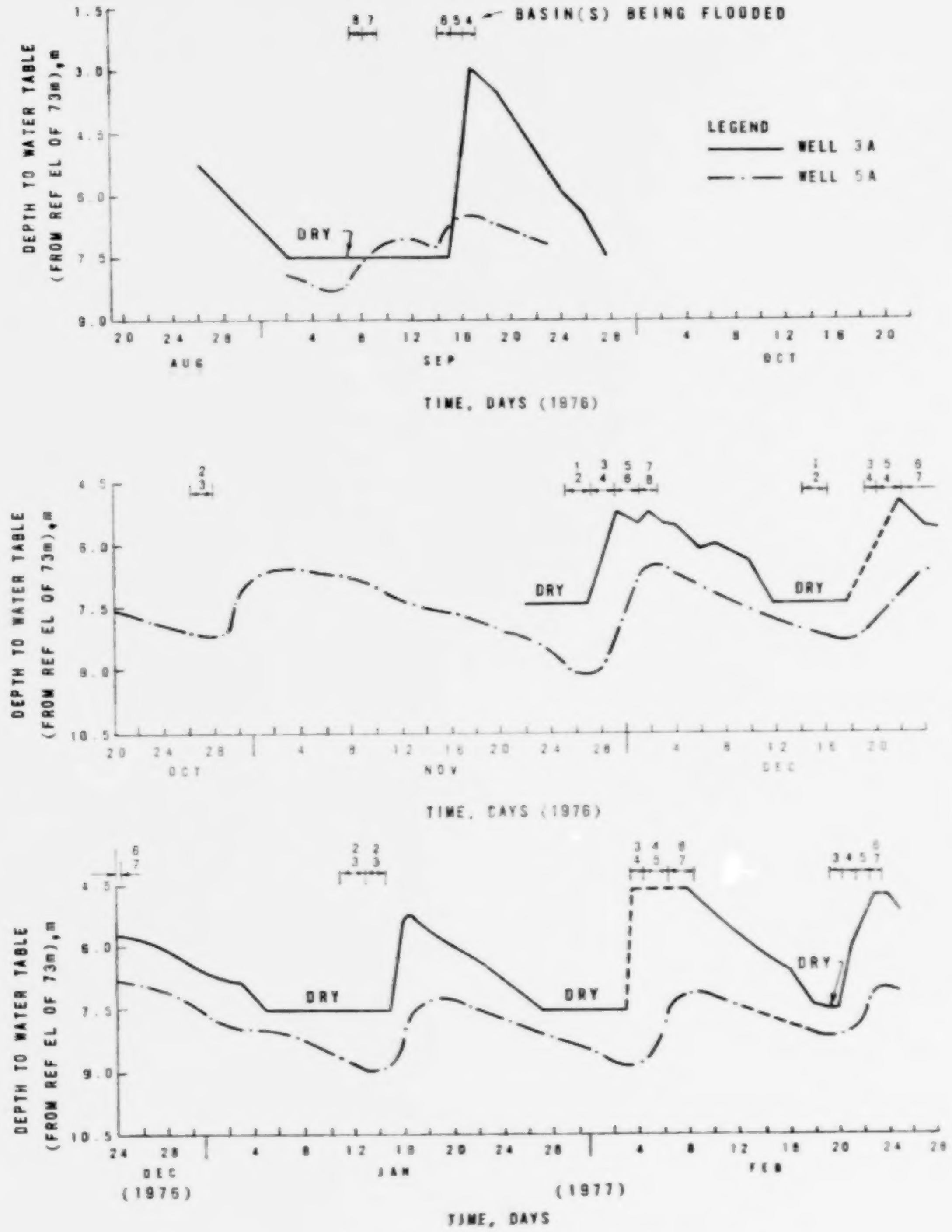

Figure 13. Water level response to basin flooding in observation wells 5A and 3A.

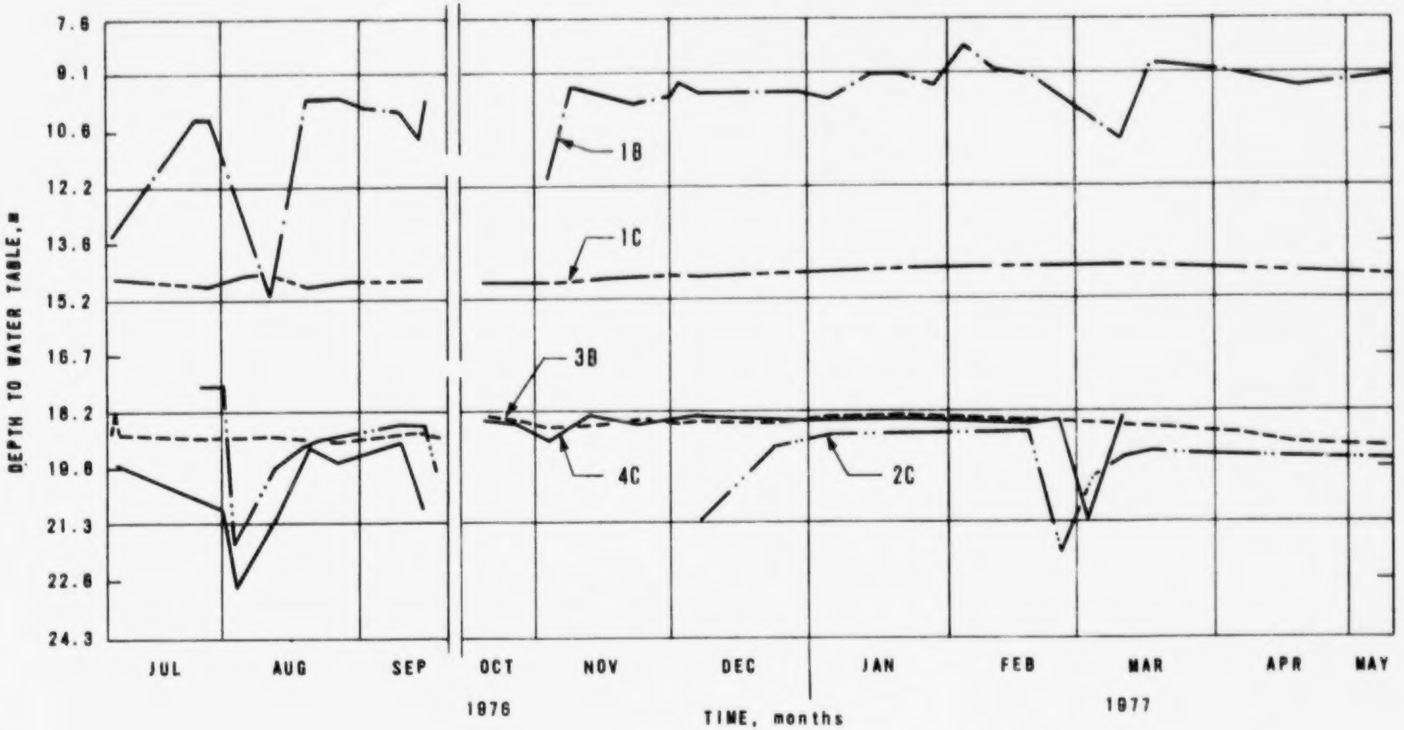

Figure 14. Water level response of intermediate and deep wells.

The deep observation wells, at Sites 1, 2, and 4, did not show any response to effluent application (Figure 14). The lower permeable zone in which these wells were screened is overlain by a 20 to 35 m (60 to 110 ft) layer of clay and silt that is probably continuous beneath the treatment site and is an effective barrier to vertical groundwater movement.

SUMMARY AND DISCUSSION

The data collected during the observation well drilling and the water level monitoring indicated that the movement of the effluent through the ground is complex and controlled by highly variable subsurface conditions. The intermediate and deep observation wells showed no response to effluent application in the infiltration basins. The infiltrating effluent apparently formed a mound which is perched on top of the clay layer that underlies the upper permeable zone. When infiltration ceased, the mound subsided mainly by lateral outflow from the site.

The mound that resulted from the flooding of Basins 3 through 8 was dissipated by lateral outflow to the north. The hydrograph of Well 3A indicates that Site 3 is in a more permeable part of the upper zone, which is probably a preferred path for infiltration and outflow. The exit paths for mounds created by flooding other basins are not known, although it is probable that the outflow was to the north for Basins 1 through 13 and toward the south or southeast for Basins 14 through 20. No lateral outflow was observed at Site 1, southwest of the treatment facility.

Some vertical leakage may be occurring also, especially near Site 3 where the clay and silt between the upper and intermediate permeable zones is the thinest. A comparison of iron and manganese concentration in effluent and groundwater indicates that vertical leakage from the upper to the intermediate permeable zone is taking place (see Section 9).

SECTION 6

SOIL INVESTIGATION

INTRODUCTION

The Hollister rapid infiltration site is located in an area mapped as a Metz sandy loam [3], though the terms gravelly sand or gravelly sandy loam appear more appropriate. The soil is alluvial, currently classified as a Typic Xerorthent [3]. A representative profile is shown below [3]:

Ap 0 to 20 cm (0 to 8 in.), grayish-brown (2.5 Y 5/2) sandy loam, very dark grayish-brown (2.5 Y 3/2) when moist; weak, fine granular structure; soft when dry, very friable when moist, nonsticky and nonplastic when wet; few very fine and fine roots; few very fine and fine tubular pores; mildly alkaline; slightly effervescent; clear, smooth boundary.

A1 20 to 35 cm (8 to 14 in.), grayish-brown (2.5 Y 5/2) sandy loam, very dark grayish-brown (2.5 Y 3/2) when moist; weak, fine subangular blocky structure; soft when dry, very friable when moist, nonsticky and nonplastic when wet; few very fine and fine roots; few very fine and fine interstitial pores; moderately alkaline, slightly effervescent; clear, wavy boundary.

C1 35 to 108 cm (14 to 43 in.), light brownish-gray (2.5 Y 6/2) sand, grayish-brown (2.5 Y 5/2) when moist; single grain; loose when dry or moist; nonsticky and nonplastic when wet; mildly alkaline, slightly effervescent; abrupt, smooth boundary.

C2 108 to 150 cm (43 to 60 in.), light brownish-gray (2.5 Y 6/2) stratified coarse sand and gravel, grayish brown (2.5 Y 5/2) when moist; single grain; loose when dry or moist, nonsticky and nonplastic when wet; contains small lenses of silt and clay; mildly alkaline, slightly effervescent.

The site surface profile was modified by surface disking operations that were practiced to restore infiltration capacity. In addition, the soil contained more gravel and gravel lenses and generally less clay than the typical profile.

Other soil types (Nacimiento clay loam, Metz gravelly sandy loam, sandy alluvial land, river wash) are located nearby and adjacent to the site. Because soil forming processes generally do not create distinct boundaries between soil classification units and because soil properties vary within units, it was not possible to find control sites with characteristics identical to the treatment sites. The control sites finally chosen were reasonably similar to the treatment sites.

The results of physical and chemical analyses are presented separately in the following sections. The methods of analyses were discussed previously in Section 6.

PHYSICAL PROPERTIES

Particle Size

A mixture of inorganic and organic particles making up a mineral soil can be texturally classified by determining the proportions of the three major soil separates--sand, silt, and clay. An exponential rise in surface area, adsorption capacity, swelling, plasticity, cohesion, and heat of wetting occurs when soil particle size decreases from sand to colloidal clay [17]. The effect of soil texture on the direction of change of soil physicochemical properties is summarized in Figure 15.

Soil Property	Soil Texture		
	Sand	Silt	Clay
	Sand, loamy sand, sandy loam	Loam, silt loam, silt, sandy clay loam, clay loam, silty clay loam	Sandy clay, silty clay, clay
Surface Area	———	———	→
Molecular adsorption	———	———	→
CEC	———	———	→
Swelling	———	———	→
Plasticity	———	———	→
Heat of wetting	———	———	→
Water holding capacity[a]	———	———	→
Infiltration[a]	←	———	———
Percolation[a]	←	———	———
Permeability[a]	←	———	———
External drainage[a]	———	———	→
Aeration[a]	←	———	———
Organic matter content[a]	———	———	→
Structure[a]	———	———	→
Bulk density[a]	———	———	→
Porosity (total)[a]	———	———	→

a. Colligative properties.

Note: Arrow indicates direction of magnitude increase.

Figure 15. Physico-chemical properties of soils as related to textural classification [18].

The particle size distribution of the five sampling sites is given in Table 13. The size limits of the soil separates are identical to those used by the U.S. Department of Agriculture.

TABLE 13. PARTICLE SIZE DISTRIBUTION IN SOIL SAMPLE FROM THE HOLLISTER RAPID INFILTRATION SITE

Depth, cm	USDA textural classification[a]	Gravel	Sand	Silt	Clay[b]
		←	mm		→
Control No.1					
0-16	Gravelly sandy loam	6	63	26	5
25-35	Gravelly sand	16	81	3	--
95-105	Gravelly sand	30	69	1	--
295-305	Gravelly sand	42	52	6	--
Control No.2					
0-16	Gravelly sandy loam	4	66	23	7
25-35	Gravelly sand	24	71	5	--
95-105	Gravelly sand	56	43	1	--
295-305	Gravelly sand	15	84	1	--
Treatment site No.1					
0-16	Gravelly loamy sand	20	67	13	--
25-35	Gravelly sand	10	86	4	--
95-105	Gravelly loamy sand	38	54	8	--
295-305	Gravelly sandy loam	6	70	17	7
Treatment site No.2					
0-16	Sandy loam	2	64	28	6
25-35	Gravelly loamy sand	10	70	20	--
95-105	Gravelly sand	30	69	1	--
295-305	Sandy loam	1	67	26	6
Treatment site No.3					
0-16	Gravelly sandy loam	6	64	26	4
25-35	Gravelly sand	8	88	4	--
95-105	Gravelly sand	29	70	1	--
295-305	Gravelly sand	15	82	3	--

a. >2 mm = gravel; 0.05-2 mm = sand; 0.002-0.05 mm = silt; <0.002 mm = clay.

b. Clay fractionation (by hydrometer) was performed only when the analyst had reason to believe that clay was present in the pan sieve sample. When clay analysis was not performed, the soil was assumed to be all silt.

mm x 0.03937 = in.

Control sites are either gravelly sands or gravelly sandy loams while the treatment site textures range from gravelly sand to gravelly sandy loam. In all cases, sand is more than 50% of the less than 2 mm size fraction. Because soils in this textural class have large spaces between particles, they are expected to have a high percolation rate, low water holding capacity, and high aeration capacity [18]. Field inspection of the sand fractions from both control and treatment site locations indicated that they consisted largely of quartz, feldspar, and shale fragments.

Surface Area

It has been shown that for a specialized set of conditions and spherical particles, an estimate of soil surface area can be obtained using the relationship [19]:

$$\frac{A}{M} = \frac{3}{\rho r} \tag{1}$$

where A = surface area
M = mass of soil
ρ = particle density
r = particle radius

To calculate rough values of surface area per unit mass, a particle density of 2.65 g/cm^3 was selected, while particle radii for each size fraction were calculated from the average particle diameters given in Table 14.

TABLE 14. SURFACE AREA PER UNIT SOIL MASS

Size fraction	Gravel	Sand	Silt	Clay
Average diameter, cm	0.30	0.0102	0.0026	0.0001
A/M = 3/ρr, cm^2/g	8	22	870	22,600

cm x 0.3937 = in.
cm^2/g x 70 = in.^{2}/lb

Estimates of surface area distribution for a hypothetical 100 g soil sample are presented in Table 15. While the numbers are gross estimates only, they are qualitatively important because many soil-solution interactions, especially heavy metal sorption, are surface area dependent.

Surface area estimates range from a low of 2,264 cm^2 (351 in.^{2}) to a high of 179,694 cm^2 (27,853 in.^{2}) with surface soil horizons generally having the highest surface area. Treatment site 1, however, has its surface area maximum at 300 cm (118 in.). (Clay analysis was not performed on the surface depth fraction; a small amount of clay would significantly alter this estimate.) A similar subsurface maximum exists for treatment site 2.

TABLE 15. ESTIMATED SURFACE AREA DISTRIBUTION (cm^2) FOR A 100 g SAMPLE OF SOIL

Depth, cm	Gravel	Sand	Silt	Clay	Total
Control No. 1					
0-16	48	1,386	22,620	113,000	137,054
25-35	128	1,782	2,610	—	4,520
95-105	240	1,518	870	—	2,628
295-305	336	1,144	5,220	—	6,700
Control No. 2					
0-16	32	1,452	20,010	158,200	179,694
25-35	192	1,562	4,350	—	6,104
95-105	448	946	870	—	2,264
295-305	120	1,848	870	—	2,838
Treatment site No. 1					
0-16	160	1,474	11,310	—	15,782
25-35	80	1,892	3,480	—	5,452
95-105	304	1,188	6,960	—	8,452
295-305	48	1,540	14,790	158,200	174,578
Treatment site No. 2					
0-16	16	1,408	24,360	135,600	161,384
25-35	80	1,540	17,400	—	19,020
95-105	240	1,518	870	—	2,628
295-305	8	1,474	22,620	135,600	159,702
Treatment site No. 3					
0-16	48	1,408	22,620	90,400	114,476
25-35	64	1,936	3,480	—	5,480
95-105	232	1,540	870	—	2,642
295-305	120	1,804	2,610	—	4,534

$1\ cm^2 = 0.155\ in.^2$

Bulk Density

Bulk density measurements reflect the status of soil structure, which in turn affects water movement, aeration, and porosity. It is a measure of total soil space not occupied by solid matter and is affected by texture, organic matter content, aggregation, root penetration, and compaction. Bulk density measurements were performed at selected locations throughout the study area using the methods described previously. The results are presented in Table 16.

Average measurements ranged from 1.27 to 1.72 g/cm^3, but there were not sufficient data to determine if conclusive differences existed between control and treatment sites or among depths.

TABLE 16. SOIL BULK DENSITY MEASUREMENTS (g/cm^3) AND CALCULATED SOIL POROSITIES

Site	Depth, cm	D_b Core method	D_b Excavation method	D_b Average	% Porosity
Control No. 1	0-8	1.59	1.84	1.72	35
	25-33	1.65	—	1.65	38
Control No. 2	0-8	1.41	—	1.41	47
	25-33	1.27	—	1.27	52
	95-103	1.57	—	1.57	41
Treatment site No. 3	0-8	1.25	1.33	1.29	51
	4-12	1.38	—	1.38	48
	25-33	1.57	—	1.50	42
	95-103	1.48	—	1.48	44

g/cm^3 x 27.8 = lb/in.^{3}

Soil porosity can be calculated from bulk density as follows:

$$\%P = \left(1 - \frac{D_b}{\rho}\right) 100 \tag{2}$$

where %P = percent pore volume
D_b = bulk density
ρ = particle density, assumed as 2.65 g/cm^3

The results of such calculations are also presented in Table 16.

Control site 2 and treatment site 3 are similar, having porosities in the range 42 to 52%. Control site 1 is slightly lower with an average porosity of 37%. The lower than average bulk densities of the near-surface measurements at the treatment site may have been the result of disking operation.

Infiltration Studies

Water intake rates were measured for several basins where wastewater had been applied for 30 years. The results of the basin flooding infiltration studies are presented in Table 17.

The average infiltration rate for the basins was 17.7 cm/d (0.58 ft/d) which is over twice the average daily application rate of 8 cm/d (0.27 ft/d). Basin 11 was tested without prior disking, and may have caused larger than normal pore clogging and the subsequent low infiltration rate.

TABLE 17. BASIN FLOODING INFILTRATION RATES

Location	Infiltration rate, cm/d
Basin 3	17.7
Basin 4	17.7
Basin 5	20.4
Basin 6	14.6
Basin 7	20.7
Basin 9	18.3
Basin 10	8.8
Basin 11	4.6
Basin 12	22.0
Basin 13	31.7
Average	17.7

cm/d x 0.3937 = in./d

Infiltration rates are generally thought to be a function of many factors, including initial water content and temperature which were not controlled in the measurement of the infiltration rate. Nonetheless, at this rapid infiltration site, infiltration rates were apparently related to soil porosity, being highest where porosity was greatest.

Cylinder infiltrometer tests were also conducted in an infiltration basin to determine the effect of wastewater solids and surface scarification on infiltration rate. Tests were conducted using clear water (SS = 0 mg/L) and primary effluent (SS = 275 mg/L) on the soil of infiltration Basin 6, which had not been disked since the last wastewater application. Tests were repeated on the same soil after disking. The testing procedure is described in reference [20] where buffer cylinders approximately 90 cm (36 in.) in diameter were continuously flooded in an attempt to approximate the true vertical infiltration rate. Inner test cylinders were 55 gal drum sections 46 cm (18 in.) in diameter. A typical test cylinder is shown in Figure 16. The infiltration rate was measured until steady state conditions were established (generally in less than 2 hours).

A one-way analysis of variance was performed on the infiltrometer data to determine if mean infiltration rates were statistically equal for the nine treatments considered (Table 18). The hypothesis of equality was rejected at the $\alpha = 0.01$ level.*

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* $\alpha = 0.01$ level is the probability of rejecting a true hypothesis.

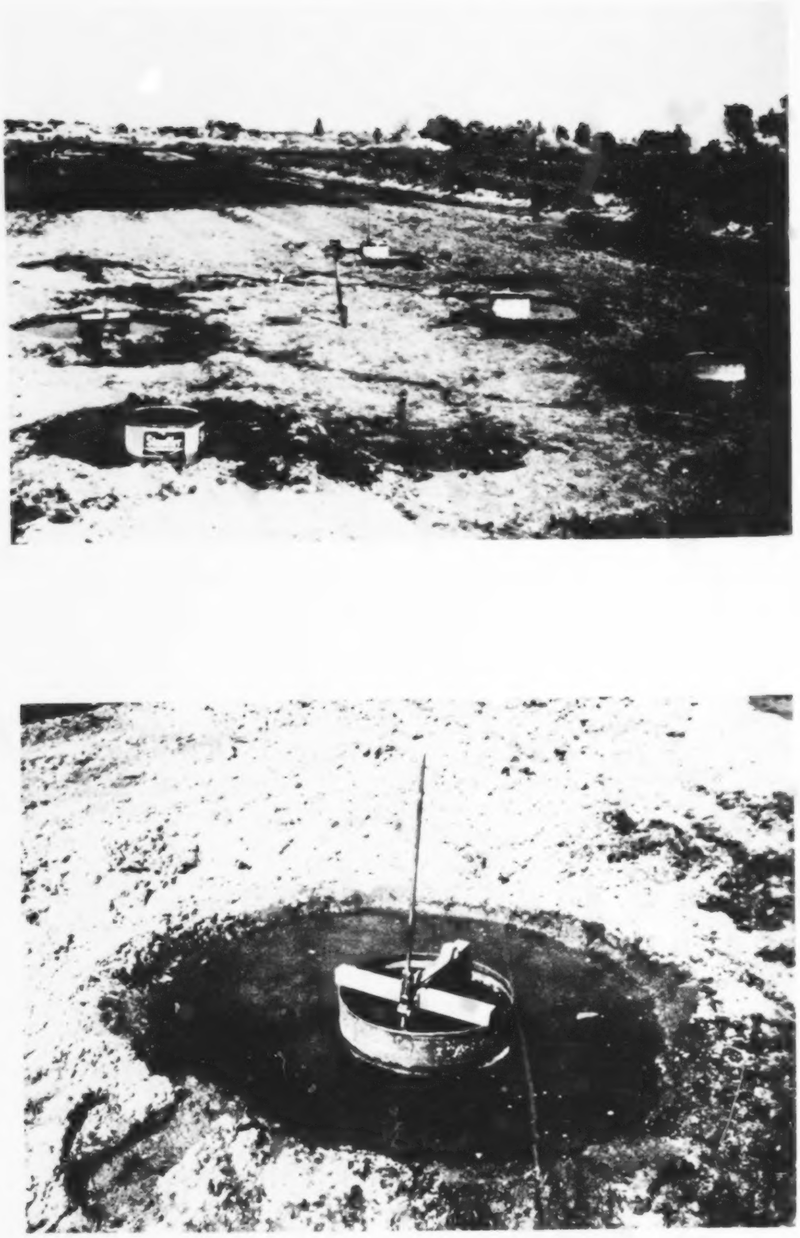


Figure 16. Cylinder infiltrometers.

Mean infiltration rates, standard deviations, and coefficients of variation are listed in Table 18.

TABLE 18. CYLINDER INFILTROMETER INFILTRATION RATES

Rank order of treatments	Location	Soil condition	Fluid	Buffer pond	Replicates	Mean infiltration rate, cm/d	Standard deviation, cm/d	Coefficient of variation, %
1	Basin 6	Undisturbed	Clear water	Yes	5	110	24	22
2	Basin 6	Undisturbed	Clear water	Yes	5	116	49	42
3	Basin 6	Undisturbed	Clear water	No[a]	5	146	30	21
4	Basin 6	Undisturbed	Effluent	Yes	6	152	73	48
5[b]	Basin 6	Undisturbed	Clear water	Yes	6	177	104	59
6	Control site	Undisturbed	Clear water	Yes	2	177	55	31
7	Basin 6	Disked	Clear water	Yes	6	268	85	32
8	Basin 6	Disked	Effluent	Yes	6	287	98	34
9	Control site	Disked	Clear water	Yes	4	311	159	51

a. Cylinders were tested (in place) after treatment No.2 without a buffer pond.

b. Cylinders were tested (in place) after treatment No.4 with clear water.

cm x 0.3937 = in.

Duncan's multiple range test, $\alpha = 0.05$ (probability of rejecting a true hypothesis), indicated significant differences between the following treatment means referred to in Table 18: 7-6, 7-5, 7-4, 7-3, 7-2, 9-6, 9-5, 9-3, 9-2, 9-1, 8-6, 8-5, 8-3, 8-2, and 8-1.

Final infiltration rates in all cases were several times the rates determined by flooding the entire basin with wastewater (see Table 17). This phenomenon was predicted by Bouwer [21] who suggested that a small buffer area in relation to inner test cylinder area may result in an overestimation of infiltration rate by cylinder infiltrometers. On the other hand, Burgy and Luthin [22] reported single cylinder infiltrometer results within 30% of the mean infiltration rate determined by flooding 12 m by 27 m (40 ft by 90 ft) plots. No difference was reported when buffer ponds were used. In the second case, test cylinders were small, (15 cm [6 in.]) and buffer cylinders were not large, (30 cm [12 in.]), relative to inner cylinder diameters.

The coarse texture of the Hollister soil may account for the high measured infiltration rates. The relatively small cylinder area as compared to that of the infiltration basin tends to decrease the effect of gas binding and soil pore clogging by wastewater solids. In addition, there was no significant difference between tests conducted with and without buffer cylinders at Hollister. This suggests that radial movement of water below the infiltrometers was occurring.

Infiltrometer results did not show a significant difference ($\alpha = 0.05$) between tests using clear water and tests with wastewater. This relation held for both undisturbed or disked soil conditions. The documented pore clogging effect of wastewater solids [23] was somehow masked by infiltrometer test conditions.

The positive effect of disking on infiltration velocity is clearly indicated by the data. Although the solids mat, which characteristically forms during flooding at Hollister, had dried and formed large cracks, the infiltration capacity was not as great as tests conducted on disked soil. Also, under similar test conditions, there was no significant difference between the control site, where wastewater had never been applied, and Basin 6 where wastewater has been applied for 30 years.

SOIL CHEMISTRY

Introduction

The soil chemistry program at the Hollister rapid infiltration site involved the analysis of 23 soil parameters. The methods of analyses have been described previously. A summary of the results is presented in Table 19.

Parameters are discussed separately or in natural groups. The data were also plotted as a function of sampling depth. For the purposes of plotting, the means are bracketed by an error bar of plus or minus one standard deviation. To facilitate plotting, treatment site points were depressed 6 cm (2.2 in.) from their true location. In all cases, squares represent control sites, and circles represent treatment sites.

For several parameters, concentration "pseudo-minimums" occurred at 100 cm (39 in.) depth. This occurred because the data were calculated on a "whole soil basis," though the analytical measurements were performed on the less than 2 mm size fraction. The larger concentrations of gravel at 100 cm (39 in.) caused the "whole soil" concentrations to appear small.

The reader is advised that the sum of the exchangeable cations Ca, Mg, Na, and K differs significantly from the cation exchange capacity (CEC). This was probably due to analytical error. Chapman [24] notes that in calcareous soils, CEC tends to be underestimated, while exchangeable Ca is overestimated. The U.S. Salinity Laboratory [12] recommends that Ca not be reported in alkaline soils while others [25] urge caution in interpretation of such measurements.

TABLE 19. RESULTS OF SOIL CHEMICAL ANALYSES

Depth, cm	pH	$CaCO_3$ equiv, %	Conductivity, µmhos/cm	Organic matter, %	Nitrogen, ppm		Phosphorus		Boron, ppm
					Total	Organic	Extractable, ppm	Total-P, ppm	
Control No. 1									
0-16	8.5	1.55	740	0.76	560	550	13	710	<0.19[a]
25-35	9.2	2.27	100	0.08	76	62	0.67	610	<0.17
95-105	9.4	2.47	100	0.02	64	45	0.56	390	<0.14
295-305	9.4	2.51	400	0.01	45	34	3.7	330	0.13
Control No. 2									
0-16	7.8	1.08	770	1.35	840	818	8.4	790	0.22
25-35	8.4	2.06	500	0.15	110	98	0.15	460	<0.15
95-105	8.9	2.27	330	0.02	34	30	0.09	360	<0.09
295-305	9.1	3.04	500	0.02	180	167	0.17	470	<0.17
Treatment Site No. 1									
0-16	6.3	0.28	2940	1.20	1200	1100	57	1600	0.66
25-35	6.8	0.27	2400	0.20	370	330	61	1200	0.36
95-105	7.9	1.17	1300	0.06	120	99	42	720	0.18
295-305	8.8	1.42	700	0.11	180	140	54	960	0.38
Treatment Site No. 2									
0-16	6.4	0.36	2620	2.19	1720	1600	109	2000	1.08
25-35	6.5	0.29	1280	0.12	350	290	73	1400	0.35
95-105	7.8	2.01	1260	0.03	93	74	39	790	0.17
295-305	8.8	1.39	880	0.08	190	150	32	840	0.47
Treatment Site No. 3									
0-16	6.7	Nm	2470	2.02	1600	1500	110	2800	0.70
25-35	7.6	Nm	770	0.28	200	174	63	1400	0.34
95-105	8.0	Nm	500	0.05	104	86	33	920	0.16
295-305	8.6	2.32	600	0.03	77	65	21	850	0.20

CONTINUED

TABLE 19 (CONTINUED)

Depth, cm	CEC, meq/100 g	Exchangeable cations, meq/100 g				DTPA extractable heavy metals, ppm								
		Na	K	Ca	Mg	Cd	Co	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Control No. 1														
0-16	7.2	0.93	0.26	5.6	1.8	0.08	0.08	<0.08	2.4	19	5.4	0.62	5.1	0.98
25-35	2.8	0.11	0.02	3.0	1.1	0.01	0.07	<0.07	0.35	7.6	2.5	0.24	0.17	0.12
95-105	1.9	Nm	Nm	1.8	1.3	0.01	Nm	<0.06	0.71	6.3	1.7	0.20	<0.14	0.04
295-305	2.3	0.12	0.01	1.5	1.5	0.01	0.01	<0.05	0.24	7.0	2.1	0.19	<0.12	0.05
Control No. 2														
0-16	8.6	0.41	0.30	5.7	1.8	0.12	0.04	<0.08	4.0	23	5.2	0.73	10.	1.6
25-35	3.3	0.07	0.05	1.8	1.1	0.02	Nm	<0.06	0.36	7.0	2.6	0.27	0.36	0.21
95-105	1.2	0.02	Nm	0.88	0.70	0	Nm	<0.04	0.17	3.7	1.2	0.14	<0.09	0.03
295-305	2.6	0.05	Nm	3.2	7.6	0.01	Nm	<0.07	0.37	8.0	2.4	0.24	<0.17	0.07
Treatment Site No. 1														
0-16	4.8	1.2	0.22	0.64	3.3	0.14	0.13	<0.06	6.7	120	2.2	0.74	3.5	11.
25-35	4.1	0.77	0.13	Nm	2.9	0.07	0.07	<0.07	2.7	104	1.0	0.54	0.59	3.4
95-105	3.2	0.39	0.06	2.8	2.2	0.02	0.09	<0.05	1.6	41	1.3	0.50	0.35	0.21
295-305	6.9	0.91	0.16	4.5	4.2	0.03	0.09	<0.08	2.3	26	2.2	0.28	0.41	0.23
Treatment Site No. 2														
0-16	11.	3.4	0.34	1.7	7.0	0.24	0.14	<0.08	8.2	270	3.1	2.4	5.7	8.0
25-35	6.2	0.59	0.16	0.09	4.0	0.05	0.05	<0.07	2.2	160	0.54	1.0	0.67	1.1
95-105	2.7	0.22	0.01	2.3	1.3	0.01	0.03	<0.06	0.87	53	2.1	0.39	0.29	0.24
295-305	7.5	0.74	0.16	6.2	4.2	0.01	0.04	<0.08	1.2	32	4.4	0.32	0.36	0.18
Treatment Site No. 3														
0-16	11.	1.2	0.38	0.94	6.3	0.18	0.04	<0.08	6.2	170	2.1	1.3	6.6	7.5
25-35	4.2	0.48	0.10	0.18	2.6	0.04	0.04	<0.07	1.4	140	0.66	0.33	0.28	1.4
95-105	2.0	0.26	0.09	0.57	1.4	0.01	0.03	<0.06	0.50	28	2.0	0.31	0.17	0.27
295-305	3.2	0.14	0.02	2.7	1.9	0.01	0.05	<0.07	0.60	14	4.6	0.44	0.17	0.32

a. Below detectable limits

Note: Nm = not measurable.

cm x 0.3937 = in.

Statistical Analysis of Soil Data

Three types of statistical analyses were performed for each soil species. The tests and corresponding hypotheses are presented below.

Two-way analysis of variance (ANOVA)	H_0: $\mu_{\text{controls}} = \mu_{\text{treatment site}}$
	H_0: $\mu_{\text{depth 1}} = \mu_{\text{depth 2}} = \mu_{\text{depth 3}} = \mu_{\text{depth 4}}$
	H_0: Depth-treatment interaction = 0
t-test (at each depth increment)	H_0: $\mu_{\text{controls}} = \mu_{\text{treatment site}}$
Duncan's multiple range test	H_0: paired treatment means are equal
	H_0: paired depth means are equal

In all cases, μ equals the mean of the subscripted soil variable.

The theory and assumptions underlying these statistical analyses are presented in several texts [26, 27]. Briefly, it is assumed that the hypothesis to be tested is true. Then the consequences of this assumption are examined in terms of a sampling distribution. If, as determined from the sampling distribution, observed data have a relatively high probability of occurring, the decision is made that the data do not contradict the hypothesis. On the other hand, if the probability of an observed set of data is relatively low when the hypothesis is true, the decision is that the data tend to contradict the hypothesis, and the hypothesis is rejected.

The level of significance α, of a statistical test defines the probability level that is to be considered too low to warrant support of the hypothesis being tested, that is, α is the probability of rejecting a true hypothesis. If the probability of observed data, p, is smaller than the level of significance α, then the data are said to contradict the hypothesis being tested, and a decision is made to reject the hypothesis.

The value of α was arbitrarily set at 0.10 for the two-way ANOVA, and depth increment t-test, and at $\alpha = 0.05$ for Duncan's multiple range test. Thus when the calculated value of p was less than α, the above hypotheses were rejected [28].

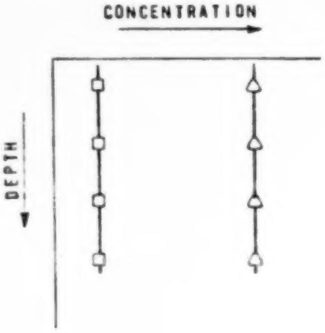
For the two-way analysis of variance, measurements for both control and treatment sites were given the following symbolic designations for statistical treatment.

	T_1	T_2
D_1	X_{111}	X_{121}
	X_{112}	X_{122}
		X_{123}
D_2	X_{211}	X_{221}
	X_{212}	X_{222}
		X_{223}
D_3	X_{311}	X_{321}
	X_{312}	X_{322}
		X_{323}
D_4	X_{411}	X_{421}
	X_{412}	X_{422}
		X_{423}

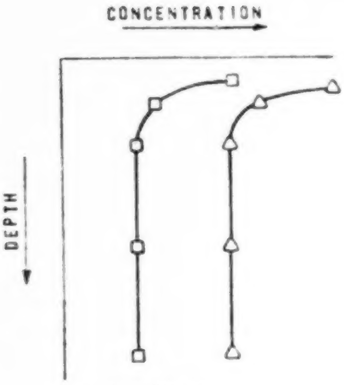
where D_1 = depth increment, 0-16 cm (0-6 in.)
D_2 = depth increment, 25-35 cm (10-14 in.)
D_3 = depth increment, 95-105 cm (37-41 in.)
D_4 = depth increment, 295-305 cm (116-120 in.)
T_1 = control site treatment
T_2 = application site treatment
X_{ijk} = k observations for D_iT_j

Once coded, Biomedical Computer Program BMDP2V, was used to generate ANOVA tables for each of the 23 soil chemical species [29]. With the aid of such a table it was then determined if significant differences in observation magnitude existed among depths, between treatments, and/or resulted from a depth-treatment interaction.

The physical significance of such a statistical treatment is illustrated by three hypothetical situations presented below. Two curves presented in each graph represent control and treatment sites.



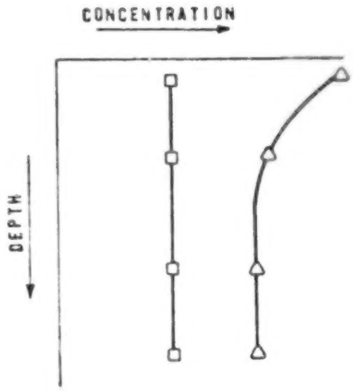
SITUATION A



SITUATION B

□ - CONTROL SITE

△ - TREATMENT SITES



SITUATION C

In Situation A, both control and treatment sites have concentrations that are uniform with respect to depth, but the treatment site exhibits a statistically significant increase in concentration relative to the control site. In contrast, Situation B shows that control and treatment sites not only exhibit a statistically significant difference due to wastewater treatment, but also shows that depth introduces a significant source of variation. Situation C introduces a third source of variation, a depth-treatment interaction. In this case, the concentration for the treatment site has increased at all depths relative to the control site. However, a selective enrichment is observed where the values of depth are low (i.e., at the surface). A statistical summary including chemical species, source of variation (D, T, DT) calculated F-statistic, and P-value, is presented in Table 20. Test statistics were deemed significant when the condition $P < \alpha = 0.10$ was satisfied.

TABLE 20. ANALYSIS OF VARIANCE FOR SELECTED SOIL PARAMETERS

Species	Source	F	P	Species	Source	F	P
CEC	D	10.02	0.001	Pb	D	29.22	0.000
	T	4.70	0.051		T	0.68	0.427
	DT	0.46	0.718		DT	1.33	0.310
Na	D	4.23	0.029	Zn	D	35.61	0.000
	T	6.51	0.025		T	38.55	0.000
	DT	0.73	0.552		DT	19.58	0.000
K	D	25.93	0.000	pH	D	20.60	0.000
	T	9.09	0.011		T	66.33	0.000
	DT	0.53	0.669		DT	3.31	0.057
Ca	D	6.10	0.009	$CaCO_3$	D	7.91	0.004
	T	4.99	0.045		T	35.54	0.000
	DT	10.68	0.001		DT	0.86	0.489
Mg	D	2.89	0.079	Conductivity	D	5.97	0.010
	T	3.20	0.099		T	27.16	0.000
	DT	1.89	0.185		DT	2.63	0.098
Cd	D	33.98	0.000	Organic Matter	D	35.11	0.000
	T	10.43	0.007		T	4.03	0.068
	DT	2.97	0.074		DT	2.29	0.130
Co	D	3.39	0.054	Total-N	D	57.82	0.000
	T	7.04	0.021		T	18.62	0.001
	DT	0.31	0.817		DT	7.89	0.004
Cr	D	6.35	0.008	Organic-N	D	66.70	0.000
	T	1.99	0.184		T	17.95	0.001
	DT	1.54	0.253		DT	8.24	0.003
Cu	D	39.94	0.000	Extractable-P	D	5.31	0.015
	T	28.56	0.000		T	66.46	0.000
	DT	4.69	0.022		DT	2.91	0.078
Fe	D	7.14	0.005	Total-P	D	9.30	0.002
	T	30.47	0.000		T	37.63	0.000
	DT	5.30	0.015		DT	2.86	0.080
Mn	D	14.17	0.000	Boron	D	9.40	0.002
	T	5.80	0.033		T	26.92	0.000
	DT	11.16	0.001		DT	5.56	0.013
Ni	D	4.66	0.022				
	T	4.88	0.047				
	DT	0.73	0.552				

In addition to the two-way ANOVA, a t-test for each depth increment and a Duncan's multiple range test were performed. These analyses helped isolate the "location(s) of heterogeneity" within the soil profile. The analyses were performed using a Hewlett-Packard combination desktop calculator-plotter. Sample output is given in Table 21. Outputs for all soil data are shown in Appendix B.

TABLE 21. SAMPLE OUTPUT FOR STATISTICAL ANALYSIS OF SOIL DATA
pH, Units

Depth, cm	Control 1 (1)	Control 2 (2)	Treatment site 1 (3)	Treatment site 2 (4)	Treatment site 3 (5)	Mean (6)	RSD[a] (7)
1. 0-16	8.500	7.800	6.300	6.400	6.700	7.140	13.528
2. 25-35	9.200	8.400	6.800	6.500	7.600	7.700	14.520
3. 95-105	9.400	8.900	7.900	7.800	8.000	8.400	8.460
4. 295-305	9.400	9.100	8.800	8.800	8.600	8.940	3.502
						8.045	
5. TOTAL	36.500	34.200	29.800	29.500	30.900	160.900	
6. MEAN	9.125	8.550	7.450	7.375	7.725		
7. RSD[a]	4.682	6.786	15.047	84.333	55.285		

	Control Average	Treatment Site Average	"t"
0-16	8.150	6.467	-5.545880
25-35	8.800	6.967	-3.537971
95-105	9.150	7.900	-6.228411
295-305	9.250	8.733	-3.661871

GROUPS OF NONHETEROGENEOUS MEANS[b]
DUNCAN'S MULTIPLE RANGE TEST.

```
8.940    8.400
8.400
7.700
7.140
  *******
9.125    8.550
8.550
7.725    7.450    7.375
```

a. Relative Standard Deviation.

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

For an $\alpha = 0.10$, the computed, absolute value of t must exceed 2.353 before the hypothesis of equality of means can be rejected. So for the case presented, treatment site and control site values of pH are significantly different at all depth increments.

pH and Calcium Carbonate

The pH of the soil-water system is critical, since it serves as a "master variable" controlling the solubility of many elements and influences microbial activity. It is important to know both the initial pH of a soil system and the expected changes in pH as the system is manipulated over time.

In a "natural system," soil pH change is affected by biological activity, organic matter, alkaline-earth carbonates, and by hydrous oxides of aluminum and iron. For an arid-zone, nonsaline soil, such as the Metz, calcium carbonate undoubtedly exhibits a major pH control.

The addition of wastewater to the treatment site has significantly decreased soil pH (Table 20). This pH reduction was apparent at all depths (Table 21). The pH differences between control and treatment sites ranged from 0.5 to 1.6 units depending on the depth below the surface. The highest average pH values were found at depths of 300 cm (118 in.), while mildly acidic conditions (pH 6.5) were found in samples from the surface of the treatment sites (Figure 17).

Simultaneously, calcium carbonate concentrations have been depleted at the treatment site. The buffering capacity of the soil was thereby diminished, permitting a pH reduction. The effect was most pronounced in the surface 30 cm (12 in.) where the difference was statistically significant.

The ultimate fate of Ca, dissolved from calcium carbonate, is uncertain as there had been no noticeable accumulation of $CaCO_3$ at depths through 300 cm (Figure 18). Although there was a slight increase in exchangeable Ca at 300 cm, it was not statistically significant. Therefore, it is possible that Ca was passing through to the underlying groundwater.

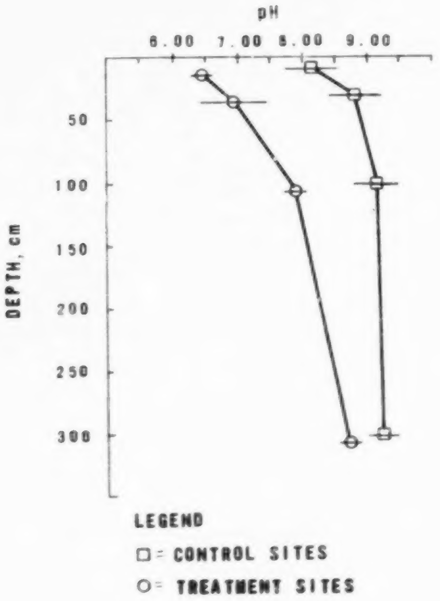


Figure 17. Vertical distribution of soil pH.

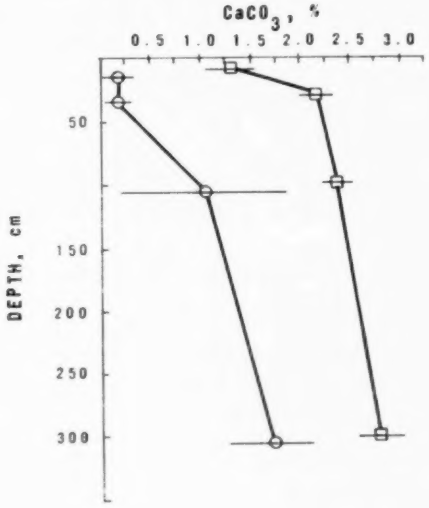


Figure 18. Vertical distribution of soil calcium carbonate.

Interestingly, the average surface soil pH at the treatment site (pH = 6.5) was less than the average pH of the applied wastewater (pH = 7.3). This suggests an additional mechanism(s) for pH reduction. One possibility is the nitrification of wastewater ammonium to nitrate according to the following reaction:

$$NH_4^+ + 2O_2 = NO_3^- + H_2O + 2H^+ \quad (3)$$

The creation of 2 moles of hydrogen for every mole of NH_4^+ oxidized would further reduce the pH of the soil system. In addition, it can be postulated that the species $CaNO_3$ would form, accelerating the leaching of Ca from the soil profile.

It should be noted as well that the pH at the control site was often in excess of that which would be predicted by equilibrating $CaCO_3$ with water (pH = 8.2). Therefore, the presence of a more soluble carbonate or bicarbonate species ($NaCO_3$, $MgHCO_3$) is suggested.

Conductivity of the Saturated Extract

Conductivity serves as a quick estimate of the amount of soluble salts in the soil system. The measured value is a function of both temperature and soil water content at the time of measurement. The USDA salinity laboratory has published several graphs and tables that allow conversion of conductivity measurements to soluble salt concentration. Input of salts at a rate greater than that removed by leaching will increase the measured conductivity. It is a valuable parameter for monitoring soil salinity and predicting plant toxicity.

A plot of average conductivity versus depth is presented in Figure 19. Both treatment and control sites show the same general trend, that is, highest extract conductivity at the surface which decreases with depth. Treatment site values are significantly higher than the controls, rising to nearly 2,700 µmhos/cm at the surface (see Table 20), reflecting the input of soluble salts in the wastewater. The interaction between depth and treatment is also significant ($p = 0.098$), as exhibited by the larger differences between surface conductivity values and those at depth.

The lack of significant difference between treatments at 30 and 100 cm (12 and 39 in.) resulted from the large standard deviations at these depths. Though not statistically significant, salinity increases may be occurring at these depths as well. At this level of soil extract conductivity, these soils could successfully grow any crop although the most salt sensitive crops might suffer some yield diminution.

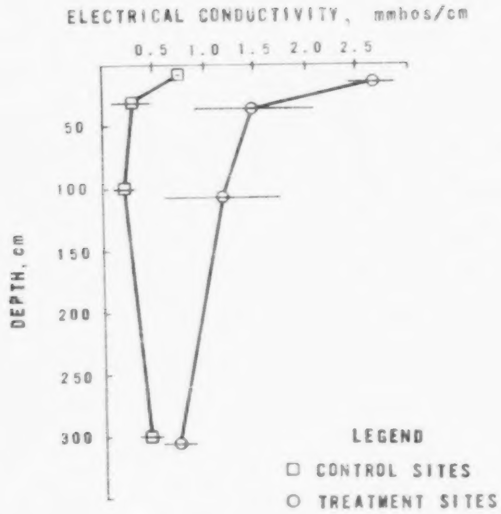


Figure 19. Vertical distribution of saturated paste, soil conductivity.

Organic Matter

Soil organic matter influences soil water holding capacity, the stability of soil structure, pH, the exchange capacity, the retention of trace metals by chelation, and the cycling of carbon, nitrogen, and phosphorus [17, 30]. It has a much higher gravimetric cation exchange capacity than any clay mineral and has a surface area comparable to that of expandable clays [18]. Its exchange capacity is highly pH dependent, being very low at pH values less than 5, but rising to over 300 meq/100g at pH values of 8.5. Of some interest is the fact that organic matter tends to counteract the unfavorable effects of high exchangeable sodium percentage on soils by stabilizing aggregates of clay minerals, thus diminishing dispersion and pore clogging. It serves as an energy material for microorganisms which in turn form humates, fulvates, and other polymers which form and stabilize aggregates.

The balance between input and decay of organic matter is modified by such factors as composition of the input material, type of microorganisms present, temperature, pH, availability of moisture, and the oxygen content in the immediate environment [17].

An analysis-of-variance (Table 20) of the Hollister soil data revealed a significant increase in soil organic matter resulting from wastewater application. Control site values range from 1.35% at the surface to 0.01% at 300 cm (Figure 20). Treatment site values range from 2.19% to 0.03% for the same depth increments.

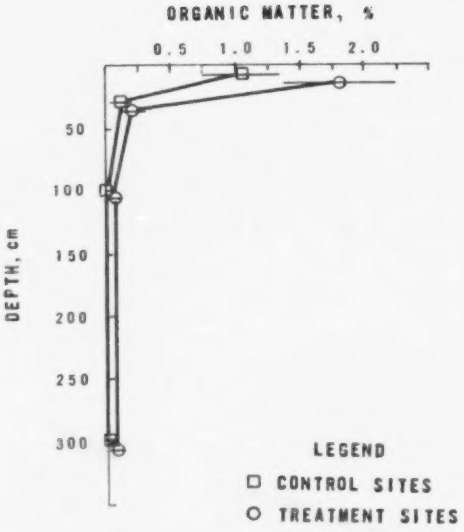


Figure 20. Vertical distribution of soil organic matter.

In contrast to the analysis-of-variance, the t-statistic revealed no contrasts between control and treatment site locations at any depth increment. This apparent discrepancy may have resulted from the smaller number of degrees of freedom used in the t-test or the assumptions implicit in the analysis-of-variance may have been violated.

Because of the above anomalies, the buildup of organic matter, if any, must be considered small. This would suggest that the soil microbial population existed in sufficient numbers to bio-oxidize most of the incoming organic matter. It should be noted, as well, that the BOD of the shallow treatment site wells was not significantly different (t-test, $\alpha = 0.10$) than the offsite control wells, further suggesting that degradation of bio-oxidizable material was occurring. The fact that COD and TOC values showed significant increases relative to the control wells suggests that some "refractory" organics were passing through the soil profile.

Nitrogen

The nitrogen cycle in soils is quite complex. When presented in a grossly simplified manner, it can be viewed as follows: plants acquire nitrogen as ammonium or nitrate ions and form organic nitrogen; after death, the organic nitrogen is mineralized into ammonium ion which in turn may be acquired by living plants or nitrified to nitrate. Nitrate may be acquired by plants or denitrified to either nitrous oxide or nitrogen gases.

Nitrogen accumulation can be expected to occur when organic forms of nitrogen are introduced into the soil system, or when the dominant form of nitrogen is the ammonium cation [18]. In contrast, nitrogen losses are expected when nitrogen is in a mobile anionic form (NO_3^-, NO_2^-) or when it is converted to a gaseous species (N_2, N_2O). The former loss is by leaching, while the latter loss is by atmospheric escape and leaching [23].

The rates of nitrogen conversion, and therefore rates of accumulation and losses, are influenced by pH, temperature, aeration, moisture, availability of a carbon source, and the presence of the appropriate microorganisms. The rate and magnitude of nitrate loss to the underlying groundwater is of some importance because it has been one of several associated causes of methemoglobinemia in infants. The USPHS has recommended that domestic water supplies contain less than 10 ppm nitrate-N.

Significant concentration increases resulting from wastewater treatment were observed at the Hollister site for the two nitrogen species measured--total and organic-N. Differences were significant for the first three depth increments, but nonsignificant at 300 cm (118 in.). The results are plotted in Figures 21 and 22, respectively.

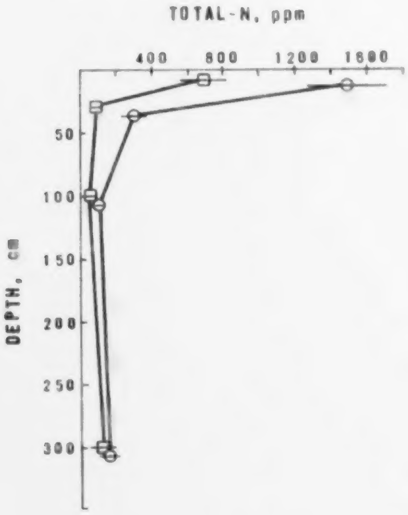


Figure 21. Vertical distribution of soil total-nitrogen.

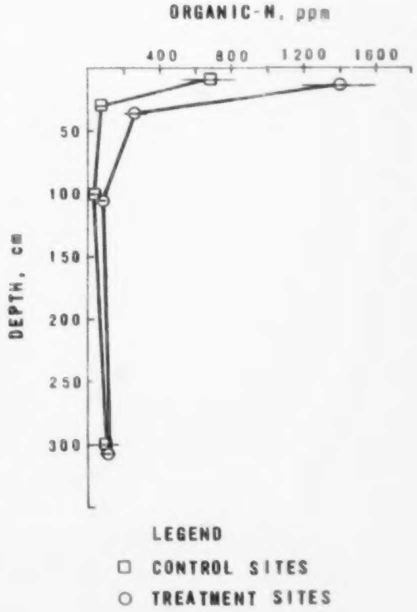


Figure 22. Vertical distribution of soil organic-nitrogen.

Total-N and organic-N were seen to be closely related in both trend with depth and magnitude. The highest concentrations of total-N occurred at the surface where values of nearly 1500 and 700 ppm were observed for treatment and control sites respectively. The values decreased by factors of one-fifth and one-tenth at 300 cm depth. Organic-N accounted for 80% or more of the total-N throughout the soil profile, though it accounted for only 40% of the total-N in the wastewater. This suggests that organic-N was being preferentially bound by the soil.

A comparision of total-N input with the mass of total soil nitrogen currently in excess of background levels (mass balance) revealed that the surface 300 cm of soil can account for only 2% of the total-N applied over the 30 year period. This implies that much of the nitrogen had been converted to mobile forms and no longer exists within the sampled soil profile. Conversion to N_2 or N_2O would result in atmospheric losses, while conversion to NO_3^- would result in losses to the underlying groundwater. Evidence for both processes is presented in the following paragraphs.

Nitrate-N levels in the shallow groundwater were higher than input levels (Section 9, Table 28) and suggest that some nitrification was occurring. However, the high background nitrate-N levels in the offsite control wells, especially at 9C, make this conclusion tentative.

The case for denitrification is presented as follows:

Organic-N is generally coverted to NO_3-N without much change in absolute concentration. If nitrification and subsequent loss to the underlying groundwater were occurring, total-N levels in the shallow aquifer would be expected to approach input total-N levels. Examination of the groundwater and effluent data (Table 28) reveals that input levels of total-N are significantly greater (t-test, $p = 0.00$) than total-N levels in the shallow groundwater. This suggests that denitrification and subsequent atmospheric nitrogen loss was occurring. In addition, the necessary pH, carbon source, and anaerobic conditions (when flooded) for such a conversion exist at the treatment site. Further evidence for the reducing conditions necessary for denitrification are given in the heavy metals section where evidence for reduction of MnO_2 to Mn^{+2} is presented.

Phosphorus

Soil phosphorus is known to occur both in organic and inorganic forms. The chemistry and thermodynamic relationships of the organic forms, however, are less well understood.

In general, the major controls exerted by the soil system on inorganic phosphorus availability are adsorption onto clay minerals and hydrous oxides of iron, aluminum, and manganese as well as the formation of cryptocrystalline precipitates with calcium, iron, and aluminum. Organic forms are thought to be adsorbed in a manner similar to inorganic forms [18]. Because the removal mechanisms are still not fully understood, the term sorption is increasingly used to describe any loss of soluble phosphorus from the system [17, 23, 31]. The extent of sorption is modified by soil type, particle size, pH, redox potential, temperature, organic matter content, and reaction time [18].

Classical adsorption isotherm expressions have been used to describe phosphorus sorption. The two most common are the Langmuir and Freundlich expressions; though the slow mineralization of phosphorus in organic matter, as well as the slow migration of adsorbed P to interfacial precipitation sites, makes it impossible to determine the precise sorption capacity of a soil [32]. Theoretical and empirical equations have been used to describe the kinetics of phosphorus uptake by soils [31].

At the Hollister rapid infiltration site, significant differences between control and treatment sites were observed ($p = 0.002$) for both total and bicarbonate extractable phosphorus at every depth increment. The significance of a depth-treatment interaction is easily observed in Figures 23 and 24. Both total and extractable P show a greater accumulation in the surface 100 cm (39 in.) of the soil profile than shown at depth. At the surface, treatment site total-P increased from 750 to 2,100 ppm while extractable-P increased from 9 to 92 ppm.

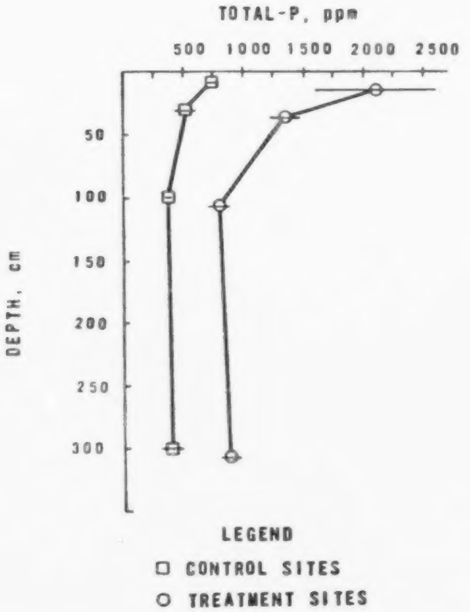


Figure 23. Vertical distribution of soil total phosphorus.

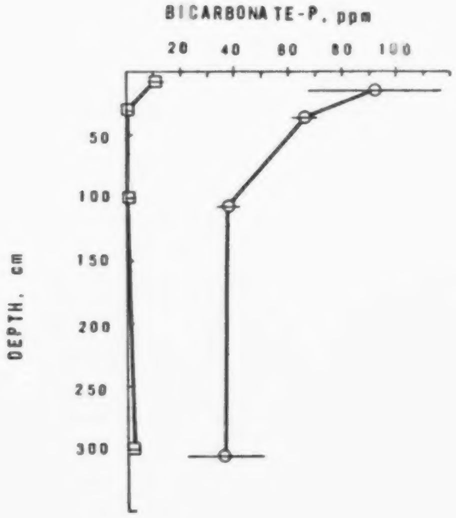


Figure 24. Vertical distribution of soil bicarbonate extractable phosphorus.

The fact that extractable-P was so much higher at all sampling depths in the treatment sites not only suggests that biological availability was high, but also that soil solution P may have been passing through the soil profile. The latter is verified by noting that shallow groundwater concentrations of total P were dramatically larger (8 and 10 mg/L) than baseline concentrations of about 0.1 mg/L. In addition, shallow well levels of total-P were approaching input levels.

Mass balance calculations, performed by comparing total-P input over the 30 year period to measured soil sorbed phosphorus, revealed that only 30% of the total P applied had been retained within the upper 300 cm (118 in.) of soil. Again, this suggests significant transport of P to the groundwater below.

In addition to the field studies, phosphorus isotherms for both surface control and surface treatment sites were generated using the nondestructive technique of Enfield and Bledsoe [10]. Freundlich isotherms were used to describe the shape of the sorption curves. In Table 22, the values K, n, and multiple -r^2 terms are presented for the following equation:

$$S = KC^n \tag{4}$$

where S = sorbed P, ppm
C = equilibrium P concentration, mg/L
K,n = constants related to the energy of sorption

TABLE 22. CALCULATED FREUNDLICH COEFFICIENTS FROM PHOSPHORUS SORPTION ISOTHERMS

Location	Time, h	K	n	r^2
Control, 0-16 cm	1	6.82	0.661	0.99
	3	14.7	0.576	0.99
	53	22.2	0.459	0.97
	120	28.6	0.478	0.93
	288	33.1	0.470	0.98
Treatment site, 0-16 cm	1	3.64	0.868	0.92
	53	7.25	0.729	0.91
	288	16.0	0.579	0.85

cm x 0.3937 = in.

The corresponding sorption curves are plotted in Figures 25 and 26. The extent of sorption is a function of time and is observed to increase as the equilibration time increases. Of particular interest, however, is the extent of sorption at the treatment site. After 288 hours, an equilibrium concentration of 20 mg/L will yield 92 ppm sorbed P at the treatment site, which compares with 135 ppm at the control site for an equivalent concentration. Thus, even after 30 years of wastewater application, the surface soil is retaining 68% of its experimentally determined sorption capacity.

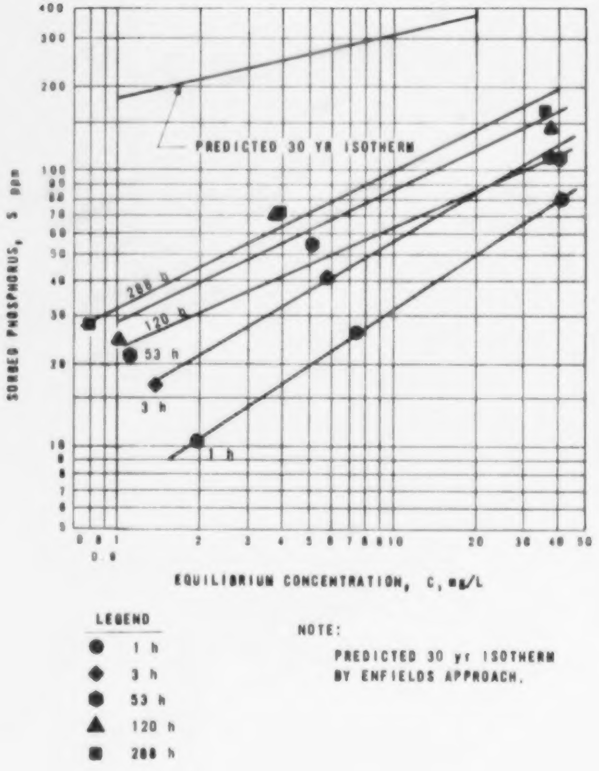


Figure 25. Logarithmic plot of Freundlich regression equation to the experimental data for sorbed phosphorus as a function of time and equilibrium concentration. Control site composite, 0-16 cm depth.

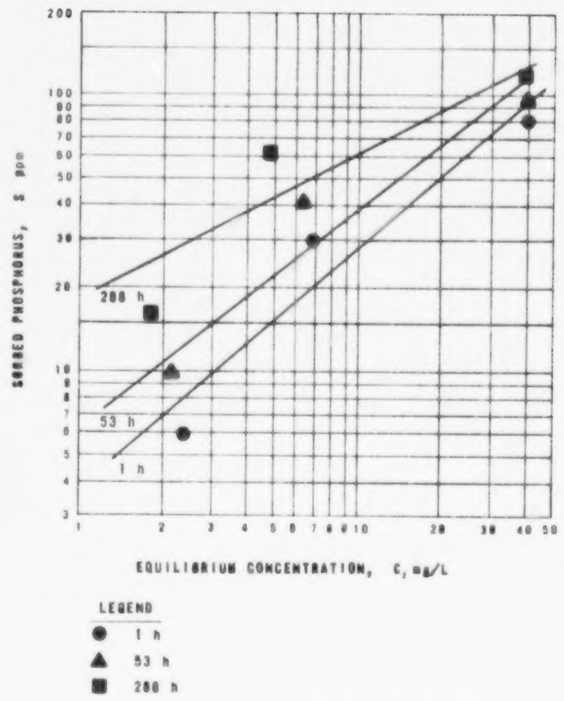


Figure 26. Logarithmic plot of Freundlich regression equation to the experimental data for sorbed phosphorus as a function of time and equilibrium concentration. Treatment site composite, 0-16 cm depth.

Sawhney and Hill [33] found equivalent behavior for Connecticut soils and suggested that alternating periods of wetting and drying may bring fresh mineral surfaces into equilibrium with the soil solution, creating new sites for P sorption. Enfield [32] notes that there is a balance between sorption and conversion of adsorbed P to more insoluble forms. He suggests that sorption is initially rapid onto mineral surfaces, but as time increases, reactions occur which utilize adsorbed orthophosphate to form phosphate minerals that have solubilities that are somewhat less than the adsorbed forms. This regenerates some sites for adsorption. Therefore, short-term laboratory P isotherm studies represent minimum estimates of field P sorption.

Tofflemire [34] approached the problem of relating laboratory P isotherms to field studies in an empirical manner. He noted that at Lake George, New York, a "mineralization factor" of 6 could be used to relate actual field P removal after 36 years of wastewater application to that predicted by a 120 hour isotherm. Further, he indicated that a factor of at least 2 would be appropriate for most rapid infiltration systems. Using the Hollister soils data and Tofflemire's approach, a "mineralization factor" of 15 is obtained (1,367/91 = 15, where 91 ppm is the amount of P sorbed after 120 hours at 12 mg/L equilibrium concentration [the average total-P input level] and 1,367 ppm is the average difference between control and treatment site total-P concentrations).

Enfield [31] and Enfield, et al. [32] took a more quantitative approach to relating laboratory studies to actual field P removal. They noted that P removal is a function of solution P concentration in the surrounding solution (C), and the amount of P already sorbed by the soil (S). Three functions are presented below:

	Model differential form	Model integrated form (S = 0, at t = 0)
1.	$\frac{\partial S}{\partial t} = \alpha(KC - S)$	$S = KC - \left[\frac{KC}{e^{\alpha t}}\right]$
2.	$\frac{\partial S}{\partial t} = \beta(mC^n - S)$	$S = mC^n - \left[\frac{mC^n}{e^{\beta t}}\right]$
3.	$\frac{\partial S}{\partial t} = a\ C^b\ S^d$	$S = \left[C^b\ (1 - d)at\right]^{1/1-d}$

If values of $\partial S/\partial t$, C, and S are known (from isothermal plots) the constants α, K, β, m, n, a, b, and d can be evaluated using multiple regression techniques. The results of such an exercise are presented in Table 23, for t in hours, C in mg/L, and S in ppm. Because concentrations greater than 20 mg/L are generally not encountered in domestic wastewater, only points below this equilibrium concentration were included when evaluating these coefficients.

TABLE 23. EVALUATION OF THREE DYNAMIC PHOSPHORUS SORPTION MODELS

Model	Coefficient	Value	r^2
1	α	0.369	0.75
	K	6.64	
2	β	0.45	
	m	22.47	
	n	0.53	
3	a	4490	0.77
	b	2.24	
	d	-3.48	

Plots of dynamically predicted versus equilibrium observed values of $\partial S/\partial t$ are presented in Figures 27 through 29, respectively. Models 1 and 2 are adequate when values of $\partial S/\partial t$ are greater than 5 ppm/h (i.e., when equilibration times are short), but fail when $\partial S/\partial t$ is small. Model 3 appears to be the least satisfactory model, and is inadequate throughout the range of $\partial S/\partial t$. Order of magnitude differences are observed.

As with most models, they should not be extrapolated beyond the experimental equilibrium times. Models 1 and 2 approach KC and mC^n, respectively, as t increases, while S approaches infinity as t increases in Model 3. Therefore, in their present form, these models cannot be used as a predictive tool for land treatment operations.

In an effort to overcome this problem, Enfield [32] introduced a graphical interpolation technique. Underlying this empirical technique is the assumption that phosphorus removal is a function of C, S, and t.

Using Enfield's approach, a predicted 30 year isotherm was constructed and graphed in Figure 25 (previously mentioned). As is observed, the amount of sorption is shown to increase markedly over that recorded at 120 hours. Previously, it was shown that the actual amount of P removal at the Hollister site was 1,367 ppm. This compares to a predicted value of 325 ppm that would be expected to be removed at an equilibrium concentration of 12 mg/L. Hence, this method underestimates the soil's removal capacity by a factor of 4. More research is necessary to relate laboratory P sorption to field studies. Factors such as pH and Ca concentration are factors that should be accounted for in any future models. Both the models of Tofflemire and Enfield have been shown to be conservative, and therefore still useful for design purposes.

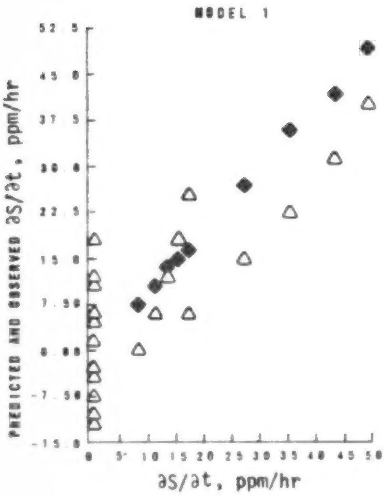


Figure 27. Dynamically predicted values versus equilibrium observed values of sorption rates (∂S/∂t). Model 1.

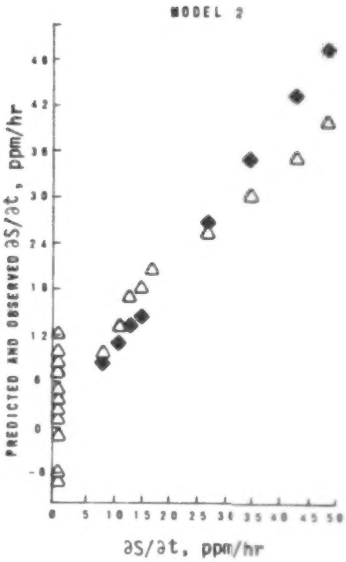
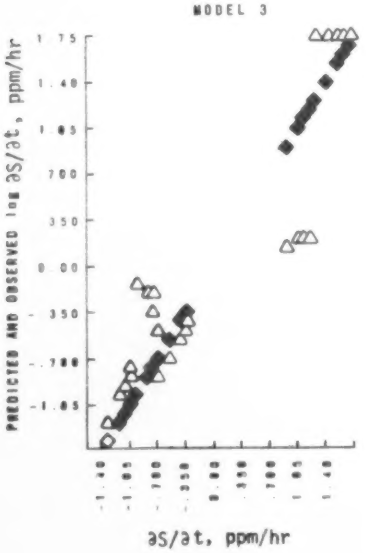


Figure 28. Dynamically predicted values versus equilibrium observed values of sorption rates (∂S/∂t). Model 2.



△ DYNAMICALLY PREDICTED

◆ EQUILIBRIUM OBSERVED

NOTE: SOME POINTS OMITTED FOR CLARITY

Figure 29. Dynamically predicted values versus equilibrium observed values of log sorption rates (∂S/∂t). Model 3

Boron

Boron is an agriculturally important species because small amounts are essential for plant growth, while even a slight excess can be toxic to some plants [36]. Boron concentrations in irrigation waters in excess of 0.5 mg/L are considered undesirable for boron sensitive crops such as citrus, stone, and pome fruit trees as well as grapes [36].

The increased use of boron compounds in certain household laundry products has tended to aggravate the problem [37]. Bingham reports that in the Santa Ana River system in California as much as 50% of the 0.75 to 1.50 mg/L boron in the sewage effluent comes from household sources [36]. Purves and Mackenzie also showed that boron in sludges and municipal waters is largely in a water soluble form [38].

It is generally assumed that the orthoboric acid species exists in nature. Since the pK of the first dissociation constant is 9.2, the undissociated form, H_3BO_3, is thought to be the dominant species in natural systems [39].

Many soils are capable of immobilizing boron. Boron adsorption occurs mainly onto the mineral fraction of the soil, though organo-boron combinations do occur [17]. Boron can be adsorbed onto iron and aluminum hydroxy compounds present as coatings on, or associated with, clay minerals [40, 41]; onto iron or aluminum oxides in the soil [42]; and onto micaceous clays although all clay minerals show some capacity for boron retention [43]. Rhoades, et al. [44] concluded that in arid soils boron adsorption is associated with magnesium hydroxy clusters or coatings on the weathering surfaces of ferromagnesium minerals such as olivine, enstatite, diopside, auguite, tremolite, and hornblende, as well as the micaceous layer silicates.

Boron adsorption occurs over a wide range of pH. Such results suggest that boron is adsorbed as boric acid under acid conditions and borate ion as the pH approaches 9.0 [41]. Boron adsorption also takes place independently of concomitant adsorption of other anions. Studies by Schalscha, et al. [45] revealed no effect from the simultaneous adsorption of sulfate or phosphate on boron adsorption. Griffin and Burau [46] found evidence for three distinct states. Two states appeared to be surface or interfacially adsorbed forms, while the third may be intracrystalline.

At the Hollister rapid infiltration site, significant differences in boron levels were observed between treatments and among depths. A significant depth-treatment interaction was observed as well, caused by the more extensive accumulation of boron at the surface of the treatment profiles than at depth (Figure 30).

While boron in the control soil is nearly uniform with depth, having a concentration of 0.16 ppm, the 0 to 16 cm (0 to 6 in.) treatment site samples had approximately four times more boron than the control samples. The differences were much less at 30 and 100 cm, but are statistically significant. Boron may be passing through to underground waters, though a significant increase in boron was not observed at 300 cm.

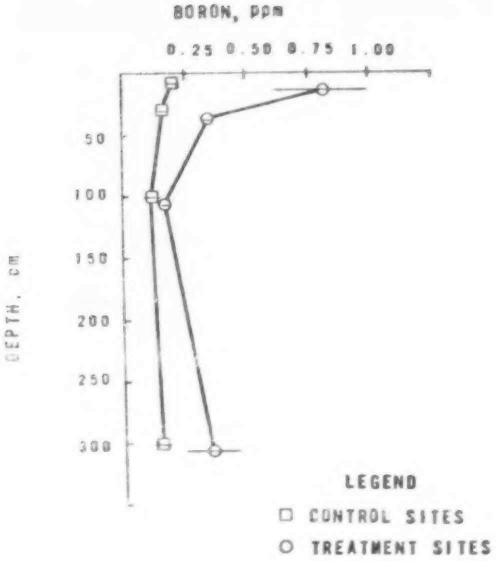


Figure 30. Vertical distribution of soil boron.

CEC and Exchangeable Cations

Electrostatic charges at soil surfaces arise from atom-proxying in aluminosilicate minerals, proton-hydroxide reactions at the aqueous-solid interface, and specific adsorption of weak acids or bases [47,48]. These negative charges create electromagnetic force fields which attract cations and repel anions. The exchanged ions near the charged surfaces are in dynamic equilibrium with the cations of dissolved salts in an outer solution. If the composition of the salt solution changes, it is reflected by a corresponding change of composition in the inner exchange phase.

The soil cation exchange capacity (CEC) is a property of paramount importance for it is associated with nutrient availability, soil permeability, and the ability of the soil to desorb cations from a wastewater influent.

Numerous equations have been developed for describing exchange reactions [48]. Most appear to have been derived from classical mass-action expressions or Donnan theory [49]. In general, the final form of these equations takes the form:

$$\frac{[A^+]}{[B^+]} = K_{AB}\left(\frac{(AX)}{(BX)}\right)^n \quad \text{for the reaction}$$

$$AX + B^+ = BX + A^+ \tag{5}$$

where the quantities in brackets represent activities of solution ions and the quantities in parentheses are exchanger concentrations [50].

The selectivity coefficient, K_{AB}, is generally not constant over a wide range of exchange compositions. Therefore, models using single-valued coefficients can only be used semiquantitatively over limited concentration ranges. This is largely due to the existence of nonequivalent bonding sites on the soil surface [47, 50]. More quantitative approaches treat the coefficient as a function of exchange composition.

The actual soil CEC is a function of pH, presence of organic matter, surface area, type of clay mineral, and the analytical method used [24]. Generally, that portion of the soil that hydroxylates or deprotonates easily as the pH is increased (amorphous hydrous oxides and organic colloids) will show a corresponding CEC increase. That fraction of the CEC due to atomic proxying will remain relatively unaffected by changes in pH. Soil surface area can be correlated with the CEC.

Affinity relationships of soil colloids for ions have been measured for many species in wastewater. In general, the selectivity of an exchanger for a given ion is a function of ionic charge and size. The preference for ions of higher charge increases with dilution of the electrolyte, while within a series of ions of equal charge, the ion with the smallest hydrated radius is preferred [49]. Therefore, for those ions generally defined as being exchangeable, the order of preference is:

$$Ca^{+2} > Mg^{+2} > K^{+} > Na^{+}$$

when the ions are of comparable concentrations.

Empirical relationships exist [12] that relate solution equilibrium concentrations of soluble cations to the concentration of exchangeable cations. The most widely used are:

$$ESR = -0.0126 + 0.01475\ (SAR) \tag{6}$$

$$EPR = 0.0360 + 0.1051\ (PAR) \tag{7}$$

where ESR = exchangeable sodium ratio

$$= \frac{\text{exchangeable sodium}}{\text{CEC - exchangeable sodium}}$$

EPR = exchangeable potassium ratio

$$= \frac{\text{exchangeable potassium}}{\text{CEC - exchangeable potassium}}$$

$$SAR = \frac{Na^{+}}{\sqrt{\dfrac{Mg^{+2} + Ca^{+2}}{2}}} \quad \text{and} \quad PAR = \frac{K^{+}}{\sqrt{\dfrac{Mg^{+2} + Ca^{+2}}{2}}}$$

where Na^{+}, K^{+}, Mg^{+2}, Ca^{+2} refer to concentrations in meq/L.

The older SAR-ESR relationships have been modified to include additional water quality parameters. Certain combinations of water soluble salts will result in the deposition or dissolution of calcium carbonate, thereby affecting the SAR. Rhoades [51] presents the equation:

$$\underset{\text{soil surface}}{ESR} = \underset{\text{irrigation water}}{SAR} [1 + (8.4 - pH_c)]^2 \tag{8}$$

Tables for calculating pH_c are presented in Ayers and Westcot [52].

In arid fine grained soils, an ESR greater than 0.15 is generally considered serious, though susceptibility to permeability degradation is a function of mineralogy, cementing materials, and other factors. Highly hydrated Na^+ replaces less hydrated Ca^{+2} and Mg^{+2}, resulting in increased dispersion of clay. This in turn adversely affects soil permeability.

Soils in the application site showed a significant increase in CEC with respect to depth and treatment. However, no depth-treatment interaction was observed. Because of the smaller number of degrees of freedom, the more specific t-statistic did not show significant CEC increases due to treatment at any depth (Figure 31). Duncan's range test indicated that CEC increases were significant only for treatment Site 2 over control Sites 1 and 2.

Control site CEC value averages ranged from 1.5 to 7.9 meq/100 g, while treatment site values ranged from 2.6 to 8.9 meq/100 g. Highest values were at the surface with lower values at depth. In spite of the pH decrease resulting from wastewater application, CEC increased. It is therefore presumed that the resulting loss of any pH dependent charge was more than offset by the influx of organic matter and mineral clays in the wastewater. As a general rule, a 1% increase in soil organic matter or clay content can result in a CEC increase of 3 or 0.8 meq/100 g, respectively.

Three exchangeable cations (Na, K, and Mg) showed significant increases in concentration after wastewater treatment. Significant differences occurred for Mg at the surface and at 30 cm (1 ft), for Na at 30 cm (1 ft) and 100 cm (39 in.), and for K at 30 cm only. No significant depth-treatment interactions were observed for any of these cations (Figures 32 through 34). This is especially apparent for Na and K whose increases appeared nearly uniform throughout the soil profile. The effect of wastewater addition on Ca (Figure 35) is confounded by the possibility of analytical error and chemical interactions and is discussed subsequently. The interpretation of Mg behavior is complicated by the inclusion of a 300 cm (10 ft) control value that appears to be anomolous (7.6 meq/100 g). Laboratory contamination appears to be the cause of this high value.

As was described previously, SAR and PAR are often used to predict ESR and EPR [48, 51]. This exercise was performed for Hollister soils. The results are presented in Table 24.

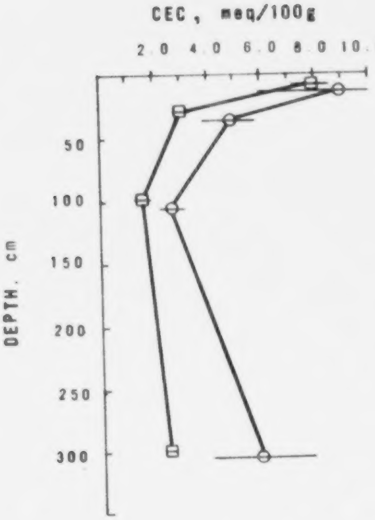


Figure 31. Vertical distribution of cation exchange capacity.

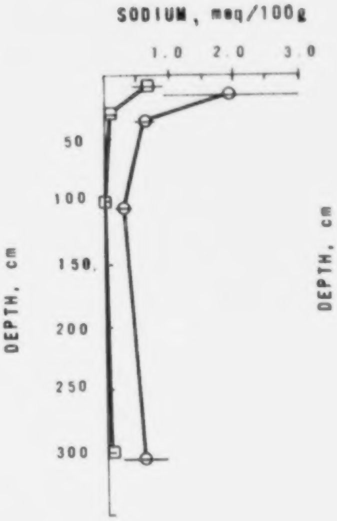


Figure 32. Vertical distribution of exchangeable sodium.

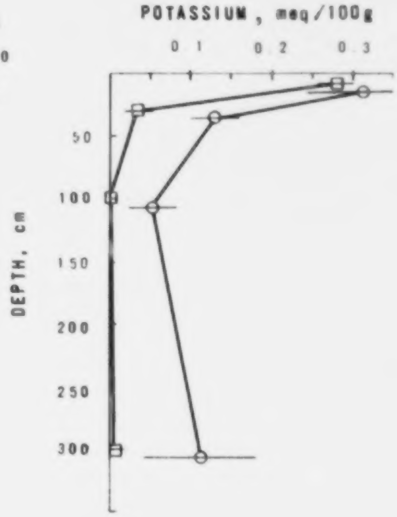


Figure 33. Vertical distribution of exchangeable potassium.

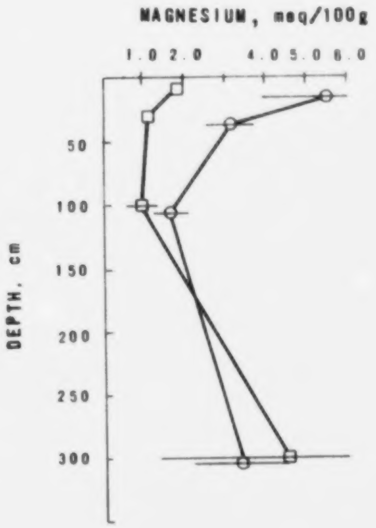


Figure 34. Vertical distribution of exchangeable magnesium.

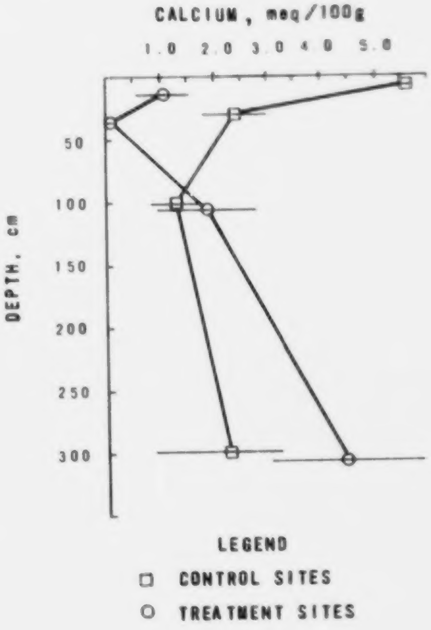


Figure 35. Vertical distribution of exchangeable calcium.

TABLE 24. MEASURED AND CALCULATED EXCHANGE RATIOS FOR SOILS AT THE HOLLISTER RAPID INFILTRATION SITE

	ESR			EPR	
	calculated		measured	calculated	measured
Location					
Treatment site	0.15[a]	0.07[b]	0.30	0.02[b]	0.04
Control site	--	--	0.10	--	0.04

a. Reference [51]

b. Reference [12]

While the results for predicted and calculated EPR are in close agreement, the data in Table 24 indicate that more exchangeable Na is present than can be predicted using either of the previously discussed functional relationships. One explanation is that the salts of Na, Ca, and Mg have been concentrated by evaporation before transport down the profile. As the absolute solution concentrations of Na, Ca, and Mg increase, the colloid preference for Na would increase. An alternative explanation is that Mg and Ca are ion-paired, complexed, or chelated to a much greater extent than Na. Sulfate, present in inorganic or organic forms (ABS, LAS), and soluble orthophosphates and organic polyphosphates are known to form complexes with both Ca and Mg. Since the composition of the exchange phase is regulated by the concentration of free ions, an SAR calculated using total concentrations would be expected to be low. Therefore, the calculated ESR underestimates the exchange phase composition when the irrigation water contains complexing species. Though ESR values are high throughout the treatment site, no apparent infiltration rate decreases have been observed.

Complications are apparent when discussing the fate of applied calcium. As mentioned previously, analytical inaccuracies arise when measuring exchangeable Ca in a calcareous soil. For this reason, the U.S. Salinity Laboratory [12] recommends that exchangeable Ca not be reported in calcareous soils. With this limitation in mind, the following discussion can only be viewed as preliminary.

Control and treatment site values for Ca are significantly different. Unlike the previous exchangeable cations, calcium decreased after wastewater addition, though the decrease was significant through the surface 30 cm (1 ft) only. A decrease of 4.5 meq/100 g Ca was observed at the surface.

The relationship of Ca to Mg is puzzling. The applied wastewater had a molar Mg/Ca ratio of approximately 2. Given that most soil colloids have nearly equal exchange affinity for Ca and Mg, it would be expected that the ionic ratio of exchangeable Mg/Ca would be approximately the same as that in the wastewater. As is observed in Table 25, the addition of wastewater increased the Mg/Ca ratio in the surface 100 cm (39 in.) to values greater than 1, but reduced the ratio at 300 cm (10 ft). This suggests that Mg was selectively replacing Ca in the surface 100 cm (39 in.) of the soil. The newly mobilized Ca traveled downward and then appeared to exchange with Mg at 300 cm (10 ft) resulting in a lower Mg/Ca ratio at 300 cm.

TABLE 25. EXCHANGEABLE Mg/Ca RATIO

Depth, cm	Control sites	Treatment sites
0-16	0.3	5.4
25-35	0.5	115.7
95-105	0.8	1.3
295-3905	1.7	0.8

cm x 0.3937 = in.

Several hypotheses can be advanced to explain this ion-selective behavior. A partial list of hypotheses is presented below. A combination of several is not ruled out.

1. Analytical discrepancies

    a. When calcium carbonate is present in a sample, reported CEC values may be too low, and exchangeable Ca too high.

    b. In treatment site samples, reported values for exchangeable Ca may be too low because orthophosphate in the acetate extract was not compensated for in flame atomic absorption spectrophotometry. Releasing agents such as Lanthanum or Strontium were not added.

2. Chemical interactions

    a. Loss of Ca in the effluent by precipitation of a calcium phosphate species in the upper regions of the profile.

    b. Selective adsorption of Mg in the upper regions of the soil profile by ligand combination. The amino-carboxylate of amino acids shows selectivity for Mg. A significant amount of this ligand would be present only if there is a sufficient amount of decomposing protein with exposed peptide linkages.

    c. Differential chelation, complexing, or ion-pairing of Ca and Mg in the solution phase. This requires a ligand(s) of high concentration that selectively chelates to or is selectively coordinated by Ca. Mg would then remain as a free ion, able to compete with Na and K for a position on the exchange phase. Organic phosphates might be one such ligand, though it is questionable whether they exist in concentrations that are sufficient to make this a complete explanation.

3. Historical

   a. Effluent composition has fluctuated over time and effluent samples are not representative.

Heavy Metals

Several sorption mechanisms have been proposed to describe soil uptake of heavy metals from a percolating solution. Specific adsorption and interfacial precipitation are thought to be the controlling mechanisms at lower or trace amounts in the soil-water system.

Bulk or noninterfacial precipitation of hydroxides, hydroxy-carbonates, sulfides, or phosphates can be a controlling mechanism at any level of a particular metal, depending on the activity or concentration of the nonmetal precipitant. At very high metal levels, cation exchange may be a controlling mechanism [35, 47, 53-57]. In some cases several processes may be operating simultaneously.

The extent of heavy metal sorption can be modified by several factors including surface area of the sorbent, charge per unit area of sorbate, pH, temperature, concentration of metals, and the presence of ion-pair complex and chelate formers [47, 49, 57]. Iron and manganese are metals that also undergo oxidation-reduction reactions and are greatly affected by biological activity [58].

The potential hazards of the heavy metals to man make them a critically important variable to monitor in any land treatment system. Where domestic wastewater is applied [59] in rapid infiltration systems, accumulation in the soil and leaching to groundwater should be assessed.

The method of heavy metal analysis was a DTPA-extraction. This procedure is thought to vary in extraction efficiency, ranging from only a few percent for iron and manganese up to 50 to 60% of the labile Cd. Therefore, unlike "total-metal" analyses, DTPA-extractable metals cannot be used for mass balance calculations, unless the efficiency of extraction for each metal is known. The DTPA method is more sensitive to changes in metal status in soil, and was therefore used for the following qualitative interpretations.

Seven of the nine monitored metals showed significant increases after wastewater treatment, with p values ranging from 0.047 to 0.000. Only lead failed to show any significant differences between treatments. Chromium was not included in any ANOVA because its concentration was below the detection limit in all cases. Depth-treatment interactions were not observed for copper, nickel, and lead.

After wastewater application, DTPA-Fe increased by nearly a factor of 9 at the surface, and was also significantly greater at other sampling depths throughout the profile (Figure 36). There is a question of whether this increase is a result of accumulation of iron by wastewater addition or the result of alternating periods of wetness and dryness, resulting in both reducing and oxidizing conditions. Reduction of ferric iron to ferrous, followed by oxidation of ferrous to ferric and subsequent precipitation as a hydrous oxide could lead to a less crystalline oxide, more soluble and more extractable by DTPA. There is no doubt, however, that iron was moving through the soil profile as shown by a relatively high iron concentration in the shallow and intermediate groundwater. There was no indication from the data whether iron was moving in the ferric or ferrous oxidation states or as complexed, chelated, or stable soil forms.

In contrast to iron, manganese was leached by the wastewater from the surface 30 cm (1 ft) (Figure 37). Surface manganese concentrations decreased from 5.3 to 2.5 ppm. Though the average increase is not significant, there seems to be some redeposition at depth, expecially in treatment Sites 2 and 3. In addition, there is evidence to suggest that manganese entered the shallow and intermediate depth wells (see Section 9). Preliminary soil analyses (using "total-Mn") indicated that not only was there a diminution in extractable manganese, but there was also significant loss of "total-Mn" in the 0 to 15 and 25 to 35 cm (0 to 6 and 10 to 14 in.) samples at the treatment sites.

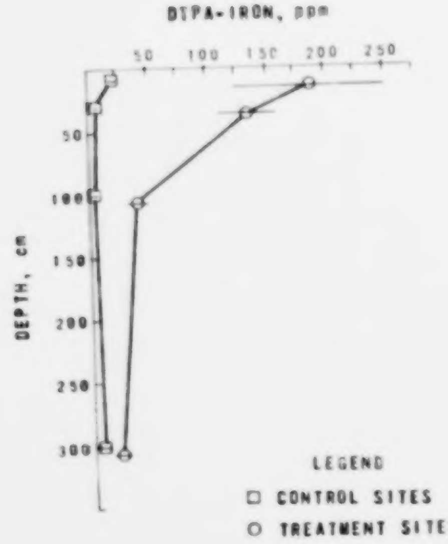


Figure 36. Vertical distribution of DTPA-extractable iron.

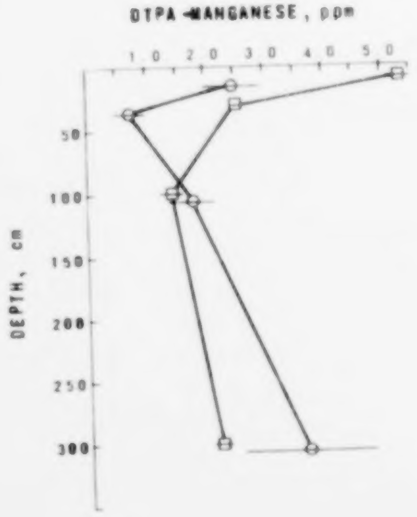


Figure 37. Vertical distribution of DTPA-extractable manganese.

One explanation of this behavior is that the influx of carbon (248 mg/L as TOC) and its subsequent oxidative metabolism by microorganisms could have created local anaerobic regions. Within these regions, there is a demand for alternative electron acceptors. Nitrate ion, if available, represents a suitable alternative to oxygen. Tetravalent manganese in soil oxides can also serve as an electron acceptor for oxidative microbiological metabolism of organic matter. The result is reduction of manganese to the more soluble divalent form, which can be lost from the surface by leaching. Redeposition would occur when and if a more oxidizing environment is encountered. Because many heavy metals are sorbed by manganese oxides, loss of these oxides from the profile may have serious consequences. Increased manganese solubilization could lead to the mobilization of formerly sorbed metals and could diminish the capacity of the soil to sorb metals entering the wastewater. Lehman and Wilson [60] and Ng and Bloomfield [61] reported similar behavior and attributed heavy metal mobilization to reducing conditions caused by fermenting organic matter.

Nickel and cobalt have similar geochemical behavior. Both tend to associate with the hydrous oxides of manganese and iron. Manganese oxides are thought to be the major control, regulating cobalt in some soils [57]. It is interesting to note that both cobalt and nickel did not increase in the surface 30 cm (1 ft) of the treatment site where manganese was lost, but did increase at 100 and 300 cm where Mn accumulated (the relationship is not statistically significant). Average DTPA-extractable cobalt was six times higher in treatment plots at 300 cm (10 ft) than in control plots at the same depth, 0.01 ppm versus 0.06 ppm (Figure 38). Similarly, average nickel values doubled from 0.17 ppm to 0.40 ppm at the 100 cm (39 in.) depth (Figure 39). Shallow groundwater concentrations of both nickel and cobalt were statistically greater than background levels measured in the offsite control wells, and suggests that both metals were passing through the soil profile.

DTPA-extractable cadmium and zinc values are summarized in Figures 40 and 41, respectively. For cadmium, wastewater application caused a significant increase only at the 30 cm (1 ft) depth where its average concentration increased from 0.015 ppm in controls to 0.053 ppm in treatment sites. The apparent increase at the surface is not significant.

DTPA-extractable zinc significantly increased in wastewater treatment profiles even at 300 cm (10 ft), where the average control value was 0.06 ppm, compared to an average treatment site value of 0.24 ppm. This suggests that some zinc passed down to and perhaps through the 300 cm (10 ft) depth. Groundwater zinc concentrations (Section 9, Table 26) were also high, further suggesting that zinc was moving downward. The relatively higher mobility of zinc contrasts with the behavior of cadmium, even though many aspects of the chemistry of zinc and cadmium are similar [62]. This apparently higher mobility of zinc may be related to its higher input concentration in the wastewater.

DTPA-extractable copper behaved in a manner similar to cadmium, showing a most pronounced accumulation at the surface, where average values increased

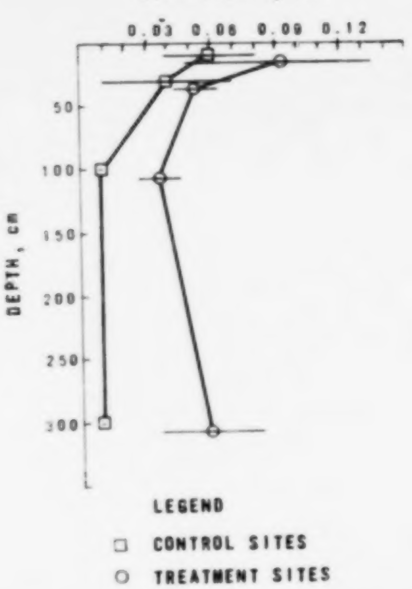


Figure 38. Vertical distribution of DTPA-extractable cobalt.

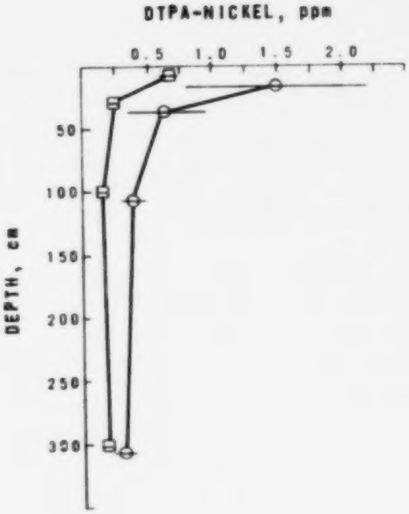


Figure 39. Vertical distribution of DTPA-extractable nickel.

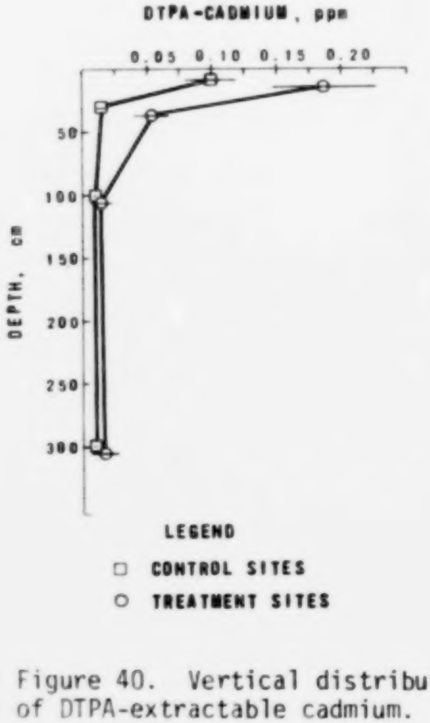


Figure 40. Vertical distribution of DTPA-extractable cadmium.

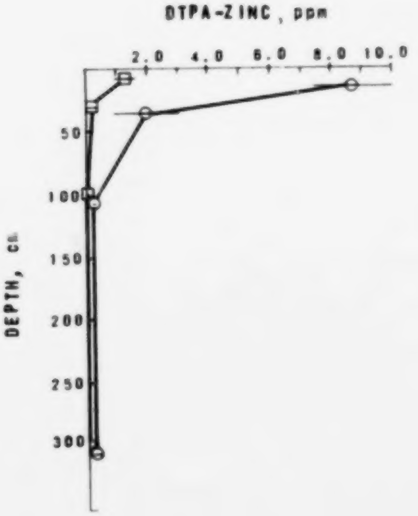
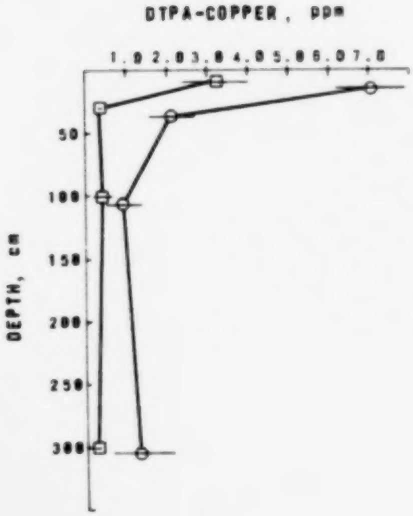
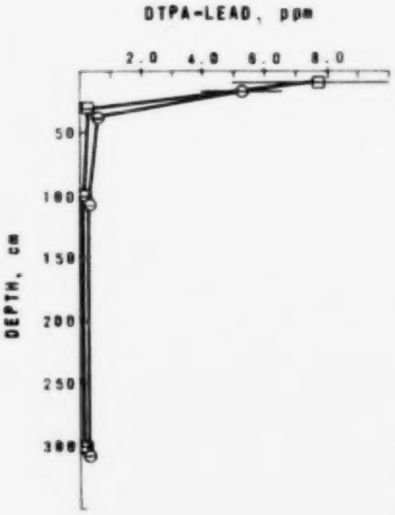


Figure 41. Vertical distribution of DTPA-extractable zinc.

significantly from 3 ppm in controls to 7 ppm in treatment sites (Figure 42). Although treatment site values were larger than controls below 30 cm (1 ft), the increase is not significant. Copper concentrations in the shallow and intermediate depth groundwater samples were nearly equal to input values. This may indicate that copper was moving unimpeded through the profile, possibly as a metal chelate or an ion pair. It may also indicate that effluent copper concentrations were not greatly different from levels in the soil-water controls.

The behavior of lead is different than any of the other metals discussed previously. Wastewater application did not cause any significant differences (Figure 43) in concentration. This may reflect the small imput of lead or it might indicate that lead is passing through the profile to the groundwater below.





LEGEND

□ CONTROL SITES

○ TREATMENT SITES

Figure 42. Vertical distribution of DTPA-extractable copper.

Figure 43. Vertical distribution of DTPA-extractable lead.

Agricultural Potential of Soils Treated with Heavy Metals

Monitoring the potential for accumulation of heavy metals within the food chain involves an evaluation of the ionic activities at the plant root surface, the amount of labile metals relative to both total composition and solution activities, and relative intensity effects in which the availability of one ion is affected by that of other ions. Plant uptake of an ion from soils is affected by inherent differences among species and varieties within species, ion interactions, and soil-plant interactions [63].

Copper, cadmium, and zinc have been the metals of primary concern in most wastewater disposal systems [64]. However, copper and zinc do not usually approach toxic levels in the ultimate food chain as a result of wastewater disposal since plant toxicities occur before these metals reach levels that would be harmful to animals and humans. Cadmium poses the greatest concern to the ultimate food chain since it is readily taken up by plants and accumulates in the vital organs of animals and humans eating these plants [64]. One way to offset this potential problem is to have in solution metals whose chemistries are similar to that of cadmium and can successfully compete for plant uptake.

Research has shown that at low cadmium levels, increasing the relative abundance of zinc with respect to cadmium can result in suppressed cadmium uptake by plants [65]. A cadmium to zinc ratio of less than 1% in the applied wastewater is suggested to guarantee protection of the ultimate food chain.

Average trace element concentrations of Hollister wastewater are listed in Table 5 (previously mentioned). While neither zinc or cadmium individually exceeds recommended maximum irrigation levels [66], the cadmium to zinc ratio exceeds 8%. It should be noted, however, that the suggested cadmium to zinc ratio of 1% is meant only for crops grown during wastewater application. Future agricultural use of the land after wastewater application, as would be the case for the Hollister site, is not addressed.

A more realistic evaluation of potential plant toxicity might be realized by evaluating the effect of soil DTPA-extractable metals on crop yields and metal uptake. Such a study was undertaken by Chaney et al. [65], though the long-term application of sludge rather than wastewater was evaluated. Ranges of DTPA-extractable metals observed in Chaney's study are shown in Table 26, together with corresponding levels at the Hollister site. The range of DTPA-extractable metal levels at Hollister is comparable to that reported by Chaney and generally falls within the low end of the range for both control and waste applied plots.

Levels of DTPA-extractable soil metals and corresponding levels of heavy metals in several crops are presented in Table 27.

TABLE 26. COMPARISON OF SOIL MICROELEMENT CONTENT AFTER LONG-TERM SLUDGE AND WASTEWATER APPLICATION

Source	Condition	Range of DTPA - extractable metals, ppm Zn	Cd	Cu	Ni
Chaney [65]	Sludge treated	1.3-95	0.10-27	1.4-100	0.2-88
	Control	2.7-7	0.08-0.16	1-4	0.2-0.6
Hollister	Wastewater treated	1.1-11	0.04-0.24	1.4-8.2	0.33-2.4
(Surface 35 cm)	Control	0.12-1.6	0.01-0.12	0.35-4.0	0.24-0.62

TABLE 27. COMPARISON OF SOIL DTPA-EXTRACTABLE HEAVY METAL WITH HEAVY METAL CONTENTS OF SELECTED CROPS [65]

		Soil				Crop				
		DTPA-extractable, air dry soil				Grains, dry crop				
Location	Soil pH	Zn	Cd	Cu	Ni	Zn	Cd	Pd	Cu	Ni
						Oat				
F-0416	6.9	8.0	1.36	4.6	0.8	30.3	0.22	0.71	4.0	2.1
F-0418	6.6	6.8	1.06	5.1	0.9	32.8	0.10	0.61	4.2	1.5
F-0420	6.4	61.8	8.4	25.3	4.1	49.8	2.13	0.56	4.7	7.5
F-0422	6.2	36.6	5.8	15.0	3.0	46.0	1.32	0.56	4.3	5.6
F-0424	6.1	25.8	3.8	10.1	2.0	45.6	1.68	0.50	4.8	5.8
						Red Clover				
F-1681	6.6	62.8	8.9	29.0	4.2	43.4	0.88	2.3	7.7	2.2
F-1682	6.6	30.1	4.5	10.8	2.3	57.6	0.92	2.4	10.8	1.6
F-1683	6.1	24.6	3.4	8.6	2.2	45.5	0.60	1.6	7.9	1.8
F-1684	5.9	24.6	3.0	7.4	2.2	50.6	0.66	2.6	7.2	2.3
						Soybean				
F-0749	6.4	8.8	0.22	5.3	0.8	48	0.30	0.50	12.0	1.8
F-0750	5.5	13.6	0.33	8.1	1.2	67	0.19	0.84	14.3	2.4
F-0751	5.9	9.6	0.23	5.2	0.6	56	0.23	0.74	12.8	2.8
F-0752	6.2	5.9	0.17	4.2	0.5	51	0.23	0.60	13.9	2.0
F-0753	6.3	3.7	0.11	2.3	0.4	45	0.18	0.89	11.8	1.5
F-0754	6.4	3.9	0.11	3.0	0.5	44	0.37	0.57	12.7	1.4
						Corn				
F-0758	5.2	7.1	0.13	3.8	1.2	19	0.23	0.87	1.8	0.7
F-0759	5.0	6.4	0.17	4.1	0.9	18	0.56	0.78	2.0	0.4
F-0760	5.0	10.2	0.22	7.1	0.8	16	0.43	0.77	1.7	0.4

Heavy metal uptake by plants appears to be a function of available metal, pH, and plant species. Uptake tends to increase as available metal increases and decreases with a rise in pH. Metal enrichment will vary from crop to crop. Though conclusions are confounded by these interacting factors, Chaney *et al.* [65] found from these and other data that the uptake of zinc, cadmium, and nickel was accelerated by sludge incorporation especially at low soil pH, and that copper and lead uptake generally was unaffected by sludge addition. Most importantly, no phytotoxicity was observed in crops ordinarily grown on farms where sludge had been used.

In light of these findings, it seems reasonable to conclude that the levels of DTPA-extractable metals will not adversely affect the future agricultural potential of the Hollister rapid infiltration site. Surface soil pH values of 6.3 to 7.6 will favor reduced metal uptake and soil DTPA-extractable metal concentrations are low relative to soils having higher metal loading rates without observed plant toxicity. Any crops grown on the site should be monitored for heavy metal uptake, however.

SECTION 9

GROUNDWATER QUALITY INVESTIGATION

INTRODUCTION

The hydrogeologic investigation determined that the shallow groundwater table beneath the rapid infiltration basins was hydraulically responsive to percolating wastewater. The shallow groundwater was sampled when underground detention time was at a minimum by taking samples at the maximum water level (See Figure 13). The results from Wells 3A and 5A, in the shallow groundwater aquifer, represent renovated water quality after 7 m (22 ft) of percolation through the unsaturated soil matrix.

The onsite intermediate wells (1B and 3B) did not show a definite water table response to effluent application (Figure 14), and groundwater quality did not convincingly indicate movement of effluent to this level. The onsite deep wells (1C and 2C), similarly, did not indicate any influence of applied wastewater on deep groundwater as observed from water table fluctuations and groundwater quality data. All results were compared to offsite control wells, upgradient and downgradient of groundwater movement in the regional groundwater aquifer.

The results of the groundwater sampling program are shown in Table 28. The values represent the arithmetic average of four grab samples taken, over the study period, according to the procedures stated in Section 6. The range, standard deviation and coefficient of variation for the individual samples are listed in Appendix C. Chemical and bacterial analyses were performed according to Standard Methods [9]. The results of effluent sampling accompany the groundwater quality results for comparison.

STATISTICAL ANALYSIS OF GROUNDWATER DATA

For purposes of analyses, well samples were grouped into four categories--shallow (A), intermediate (B), deep (C), and controls. Well 4C was originally established as an upgradient control well. The validity of this assumption was tested for every parameter. Unless there was a significant difference between Well 4C and the downgradient control wells, (at the $\alpha = 0.10$ level), it was considered to be a control well for that particular chemical parameter. Data collected over time were viewed as replicate samples.

Once properly grouped, a one-way analysis of variance (ANOVA) was performed using the SPSS statistical package [67]. Data reported as "less

TABLE 28. AVERAGE EFFLUENT AND GROUNDWATER QUALITY RESULTS
mg/L Except as Noted

			Groundwater[a]										
							Deep[d]						
	Effluent		Shallow[b]		Intermediate[c]				Offsite Control Wells				
Constituent	Range	Average[e]	3A	5A	1B	3B	1C	2C	4C	6C	7C	8C	9C
COD	546-1,029	706	46	50	21	24	18	127	16	14	14	21	30
BOD$_5$	134-414	220	6	13	3	4	5	16	3	<5	5	4	1
TOC	240-264	248[f]	10[g]	11[g]	9[g]	<1[g]	13[g]	20[g]	<1[g]	--	6[g]	--	5[g]
CCE	--	--	2.2	1.7	0.72	--	0.66	--	0.65	--	--	--	0.71
Total N	29.7-58.5	40.2	3.8	1.7	6.7	7.8	2.9	1.8	2.1	<0.2	6.0	4.1	9.9
NH_4^- - N	19.7-44.0	25.3	<0.4	<0.1	<0.2	<0.1	0.4	0.3	<0.1	<0.2	<0.2	<0.2	<0.05
N org.	6.7-21.8	14.5	2.2	1.0	0.5	0.8	2.4	1.1	<0.5	<0.2	<0.5	<0.2	0.4
NO_3^- - N	0.16-0.8	0.43	1.2	0.6	6.1	7.0	0.1	0.5	1.9	<0.05	5.6	3.9	9.5
Total P	10.0-21.5	12.4	8.0	9.6	0.1	0.2	0.1	0.1	0.1	0.04	0.03	0.1	0.1
PO_4 - P	9.0-13.2	10.5	6.8	8.7	0.1	0.1	0.1	0.03	0.04	0.04	<0.02	0.08	<0.01
Total coliforms, count/100 mL	4.6×10^6- 92×10^6	27.6×10^6	1.1×10^6	231,000	<2	1,570	9	2,160	<2	<2	<9	66	287
Fecal Coliforms, count/100 mL	3.5×10^6- 24×10^6	12.4×10^6	156,000	186,000	0	11	<1	276	0	0	<1	<1	274
pH, units	7.0-8.1	7.3	7.5	7.7	8.2	7.6	8.0	7.9	7.8	7.2	7.7	7.6	7.9
TDS	1,016-1,593	1,208	1,282	1,275	804	1,400	1,313	760	820	855	1,058	1,264	1,293
Conductivity, μmhos/cm	1,230-2,480	1,790	1,710	1,828	1,118	1,922	1,723	1,107	1,148	1,950	1,523	2,120	1,725
Alkalinity	416-465	446[f]	538	522	406	590	519	430	425	--	556	570	574
B	0.95-1.8	1.4	1.2	1.2	0.7	1.4	0.4	0.6	0.7	1.4	1.0	1.1	1.1
F	0.3-1.1	0.7	1.1	0.8	0.4	0.3	0.3	0.3	0.4	0.3	0.5	0.4	0.3
SS	206-328	274[h]	15	10	--	18	--	--	--	--	--	--	--
Na	196-391	262	249	261	139	246	207	135	112	230	162	262	225
Ca	31-71	54	100	107	46	104	56	95	47	217	115	150	80
Mg	27-76	64	71	74	81	91	118	76	89	103	95	94	109
K	10.9-16.0	12.9	13.0	14.9	3.9	9.3	5.2	5.5	4.1	4.5	3.4	5.2	5.3
Cl	185-490	284	281	292	80	295	133	81	76	250	193	282	266
SO_4	161-250	213	164	247	206	181	367	144	184	280	225	195	105
Ag	<0.005-0.012	<0.006	<0.006	<0.007	<0.004	<0.006	0.013	<0.006	<0.005	0.008	<0.005	<0.006	<0.005
As	--	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Ba	<0.1-0.24	<0.13	<0.11	<0.15	<0.11	0.12	0.11	0.11	<0.07	<0.1	<0.13	<0.06	<0.04
Cd	<0.001-0.008	<0.004	0.007	0.050	0.004	0.007	0.004	0.006	<0.003	0.003	0.02	0.007	<0.004
Co	<0.006-0.008	<0.008	<0.008	0.012	<0.007	<0.017	<0.008	<0.007	<0.006	0.008	<0.006	<0.006	<0.006
Cr	<0.004-0.036	<0.014	<0.011	<0.017	<0.006	0.013	<0.005	<0.03	0.032	<0.004	<0.004	<0.004	<0.004
Cu	0.019-0.070	0.034	0.041	0.036	0.06	0.047	0.045	0.063	0.025	0.008	<0.005	0.01	<0.018
Fe	0.14-0.82	0.39	0.21	0.51	0.46	0.86	0.04	0.21	0.03	0.02	0.14	0.08	<0.03
Hg	--	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Mn	0.055-0.094	0.070	1.30	0.62	0.22	0.54	0.21	0.10	0.01	<0.003	0.007	<0.006	<0.011
Ni	0.015-0.092	0.051	0.10	0.16	0.04	0.10	0.048	0.039	0.039	0.07	0.027	<0.05	0.026
Pb	0.012-0.12	0.054	0.10	0.08	0.05	0.08	0.044	0.067	0.012	<0.005	<0.010	<0.005	0.033
Se	--	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Zn	0.010-0.090	0.048	0.073	0.090	0.29	0.189	0.27	0.10	0.05	0.002	0.02	0.009	0.04
SAR	4.67-8.08	5.71	4.79	4.83	2.88	4.52	3.61	2.65	2.23	3.22	2.71	4.13	3.84

a. Average of 4 grab samples.

b. Depth of well screen = 7.5-10.5 m.

c. Depth of well screen = 19.5-24.0 m.

d. Depth of well screen 48 m.

e. Average of twelve 24 h composite samples.

f. Average of three 24 h composite samples.

g. Single grab sample.

h. Average of six 24 h composite samples.

cm x 0.3927 = in.

m x 3.281 = ft

than x" were input as x. The option used was "ONEWAY," which tested the hypothesis:

$$Ho: \mu_1=\mu_2=\mu_3=\mu_4=\mu_5$$

where μ_{1-4} are the means of the four groundwater groups, and μ_5 is the effluent mean.

If the hypothesis could not be rejected at the $\alpha = 0.10$ level, no further statistical analyses were performed. If rejection was warranted, pairwise contrasts (t-test) were performed to isolate the sample(s) whose mean was different. The t-statistic was computed using both pooled and separate estimates of variance. Only if Cochran's "C" statistic proved significant ($\alpha = 0.10$), indicating nonhomogeneous variance, was the latter t-statistic used. Sample output from SPSS ONEWAY is shown in Table 29.

COD, BOD, AND TOC

COD, BOD, and TOC are indicators of the strength of organic compounds present in wastewater and groundwater. The chemical oxygen demand (COD) is a measure of the oxygen equivalent of that portion of organic matter that is susceptible to oxidation by a strong chemical oxidant. The biochemical oxygen demand (BOD) test measures the oxygen uptake in the microbial oxidation of organic matter. TOC is a measure of total organic carbon. Bouwer and Chaney [68] stated that for land treatment, TOC may be a more appropriate parameter than BOD and COD, since the latter two were developed primarily for oxygen requirements in aquatic environments. TOC can also be used as a gross indicator of persistant organic compounds that remain when BOD and COD are completely removed [69]. In addition, TOC can be used as a measure of the energy (carbon) available for denitrification. Actual observed carbon requirements for effective denitrification vary in the reported literature [70, 71].

In a rapid infiltration system, the reduction of BOD, COD, and TOC is accomplished largely by the adsorption and subsequent biodegradation of organic compounds. Lance et al. demonstrated that anaerobic biological decomposition can be as effective as aerobic reactions for soil columns under conditions of continuous flooding [72] or long periods of flooding (9 days) followed by a shorter recovery period of 5 days [71].

The COD of the Hollister primary effluent ranged from 546 to 1,029 mg/L with an average concentration of 706 mg/L. The average COD concentrations in shallow Wells 3A and 5A were 46 and 50 mg/L, respectively. These values correspond to a 93% COD reduction after vertical percolation through 7 m (22 ft) of soil.

Relative to the other deep wells, COD levels in Well 2C were higher by a factor of 5 to 8 throughout the study period. This resulted from well contamination. An organic mud which was used to maintain the borehole during well construction was observed in each groundwater sample from Well 2C. When Well 2C is excluded, significant differences between intermediate, deep, and offsite control wells are not observed. This supports the contention that

TABLE 29. SAMPLE OUTPUT FROM SPSS ONEWAY

```
VARIABLE: TOTAL NITROGEN

                                    ANALYSIS OF VARIANCE

         SOURCE       D.F.     SUM OF SQUARES     MEAN SQUARES      F RATIO      F PROB.
BETWEEN GROUPS          4        11661.1698        2915.2922        119.347      0.0000
WITHIN GROUPS          46         1123.6402          24.4270
TOTAL                  50        12784.8086
                                 STANDARD     STANDARD
GROUP    COUNT        MEAN      DEVIATION       ERROR      MINIMUM     MAXIMUM     95 PCT   CONF INT FOR MEAN

GRP01      10       3.3500        2.3538       0.7443       0.2000      8.7000     1.6662    TO          5.0338
GRP02       9       7.2889        4.2040       1.4013       1.3000     14.5000     4.0574    TO         10.5204
GRP03       8       2.3000        2.3821       0.8422       0.4000      7.9000     0.3085    TO          4.2915
GRP04      12       4.5250        4.3650       1.2601       0.2000     12.2000     1.7516    TO          7.2984
GRP05      12      39.8666        7.8803       2.2748      29.7000     58.3000    34.8597    TO         44.8735

TOTAL      51      12.7490       15.9905       2.2391       0.2000     58.3000     8.2516    TO         17.2464

        FIXED EFFECTS MODEL      4.9424       0.6921                             11.3559    TO         14.1421

      RANDOM EFFECTS MODEL      16.0494       7.1775                             -7.1786    TO         32.6766

CONTRAST COEFFICIENTS MATRIX

             GRP01         GRP03         GRP05
                    GRP02         GRP04
CONTRAST  1   1.0  -1.0    0.0   0.0    0.0
CONTRAST  2   1.0   0.0   -1.0   0.0    0.0
CONTRAST  3   1.0   0.0    0.0  -1.0    0.0
CONTRAST  4   1.0   0.0    0.0   0.0   -1.0
CONTRAST  5   0.0   1.0   -1.0   0.0    0.0
CONTRAST  6   0.0   1.0    0.0  -1.0    0.0
CONTRAST  7   0.0   1.0    0.0   0.0   -1.0
CONTRAST  8   0.0   0.0    1.0  -1.0    0.0
CONTRAST  9   0.0   0.0    1.0   0.0   -1.0
CONTRAST 10   0.0   0.0    0.0   1.0   -1.0

                                        POOLED VARIANCE ESTIMATE                        SEPARATE VARIANCE ESTIMATE
                            VALUE   S. ERROR     T VALUE    D.F.         T PROB.   S. ERROR      T VALUE    D.F.       T PROB.

CONTRAST 1                -3.9389     2.2709      -1.735    46.0           0.090     1.5868       -2.482    12.3         0.029
CONTRAST 2                 1.0500     2.3444       0.446    46.0           0.656     1.1240        0.934    15.1         0.365
CONTRAST 3                -1.1750     2.1162      -0.555    46.0           0.581     1.4635       -0.803    17.4         0.433
CONTRAST 4               -36.5166     2.1162     -17.256    46.0           0.000     2.3935      -15.256    13.3         0.000
CONTRAST 5                 4.9889     2.4016       2.077    46.0           0.043     1.6349        3.051    12.9         0.009
CONTRAST 6                 2.7639     2.1794       1.268    46.0           0.211     1.8845        1.467    17.7         0.160
CONTRAST 7               -32.5777     2.1794     -14.948    46.0           0.000     2.6718      -12.193    17.5         0.000
CONTRAST 8                -2.2250     2.2559      -0.986    46.0           0.329     1.5156       -1.468    17.5         0.159
CONTRAST 9               -37.5666     2.2559     -16.653    46.0           0.000     2.4257      -15.487    13.8         0.000
CONTRAST 10              -35.3416     2.0177     -17.516    46.0           0.000     2.6005      -13.590    17.2         0.000
```

COD concentrations in both the intermediate and deep aquifers are not the result of wastewater application.

The BOD of the effluent ranged from 134 to 414 mg/L and averaged 220 mg/L. The average BOD concentrations in Wells 3A and 5A were 6 and 13 mg/L, respectively. A 96% reduction in BOD was achieved. BOD levels in all aquifers were not statistically different. Shallow aquifer concentrations were indistinguishable from background concentrations. Anomolous BOD behavior was not observed for Well 2C.

The TOC was measured only during the last sampling period. The range of 240 to 264 mg/L and average of 248 mg/L was for three composite effluent samples. The TOC concentrations of Wells 3A and 5A were 10 and 11 mg/L, respectively, for one grab sample in March 1977. A 96% reduction of TOC in the shallow groundwater is indicated. All wells have statistically equal TOC concentrations suggesting that even "refractory" organic compounds have been removed to background levels. More data on the movement of refractory organic compounds are needed, however, to base any conclusions.

The COD, BOD, and TOC results indicate that effective aerobic decomposition of organic matter was occurring. The wastewater application cycle allowed for the reduction of organic constituents to background levels. Hollister removal efficiencies compare favorably with those reported for the Fort Devens, Massachusetts, rapid infiltration site having 31 years of continuous operation. At Fort Devens, primary effluent COD and BOD concentrations of 192 and 112 mg/L were reduced to 27 and 5.9 mg/L, respectively [73]. Wastewater percolates approximately 5.4 m (18 ft) through soils with physical and hydraulic characteristics similar to those at Hollister.

RESIDUAL ORGANICS (CARBON CHLOROFORM EXTRACT)

Organic compounds present in groundwater were adsorbed onto activated carbon and extracted with chloroform [74]. This procedure was developed for high quality water and the effluent was not analyzed. The large sample volume (55 litres), time consuming adsorption procedure, and costly solvent extraction limited the number of samples tested. Carbon chloroform extract (CCE) values are expressed as mg/L, based on the total weight adsorbed onto the carbon divided by the volume of water passed through the carbon sample. While no determination of removal efficiency was possible, it can be inferred from the COD, BOD, and TOC results that wastewater application has no influence on the intermediate or deep groundwater. The CCE levels of onsite and offsite intermediate and deep wells were comparable; however, inadequate sample replicates prevented statistical relationships between CCE and other organic pollution indicators.

NITROGEN

Typical medium strength domestic wastewater may contain 40 mg/L of nitrogen of which 25 mg/L is in the form of NH_4-N and 15 mg/L is in the organic form [75]. A review of several rapid infiltration projects indicates that the concentration and form of nitrogen being applied to the soil is the same after primary or secondary treatment [69, 76-78]. Ammonium and organic nitrogen applied to soils at rapid infiltration sites are almost completely converted to the nitrate form [69, 76-78]. The availability of oxygen is critical to the conversion process. Short and frequent application cycles, from 0.5 to 3 days flooding followed by 5 to 14 days before reapplication, maximize nitrification [69, 77, 78].

In addition to nitrification, denitrification is an important conversion process. Organic carbon is needed to supply energy to denitrifying bacteria. The level of organic carbon in the supplied wastewater is critical to the removal of the highly mobile nitrate ion. Lance and Whisler observed that nitrate from a percolating wastewater was removed by denitrification [79]. The denitrifying bacteria are facultative anaerobes which use nitrate-nitrogen as an electron acceptor when oxygen concentrations become very low.

In the denitrification reaction, where glucose is used as a carbon source, 3.2 grams are required for each gram of nitrogen denitrified (C:N ≈ 3.2:1). Lance and Whisler observed that stabilized municipal wastewater (C:N ≈ 1:2.3) did not contain enough unstabilized organic matter to denitrify wastewater applied to soil columns [71]. These laboratory results were verified at the Flushing Meadows rapid infiltration project [69]. On the other hand, it is very probable that a high BOD wastewater does denitrify rapidly when applied to soil. Law *et al*. reported 83 to 90% removal of total nitrogen from overland flow treatment of high BOD cannery wastes [80]. Lance also demonstrated that with the addition of 150 mg/L glucose (C:N ≈ 5:1), soil columns intermittently flooded with secondary sewage water realized a 90% nitrogen reduction [71]. Nitrogen removal decreased to 60% when the carbon concentration was 80 ppm (C:N ≈ 2.7:1) [71]. At Hollister the C:N ratio was approximately 6 to 1, a condition favoring denitrification.

In addition to the C:N ratio in wastewater, the distribution of carbon in the soil profile influences the location where denitrification might take place. At the Hollister rapid infiltration site, soil organic matter was highest near the surface and declined progressively with depth (See Table 19). Gilmore *et al*. showed that a flooded surface soil containing 0.9% total carbon denitrified added nitrate readily without organic amendments, but the subsoil containing 0.48% total organic carbon failed to denitrify unless an available organic substrate was supplied [81]. Therefore, the zone of most active denitrification is likely to be near the soil surface in spite of its proximity to the atmosphere. This has been demonstrated in field experiments by Rolston *et al*. who observed maximum rates of production of N_2O and N_2 within the top 10 cm (4 in.) [82]. Lance *et al*. [83] also observed that denitrification takes place near the soil surface. By monitoring redox potentials versus depth in sewage flooded soil columns, it was shown that a denitrification zone was established in the top 20 cm (8 in.) or less [83].

At Hollister, a 93% reduction in total nitrogen was achieved as the wastewater passed from the surface to the shallow water table. Nitrate-N levels in the shallow water table are approximately 1 ppm and therefore posed no immediate contamination problem to permanent groundwater supplies.

Denitrification is believed to be the primary removal mechanism. Favorable conditions for conversion exist at the infiltration site, including availability of an energy source, temporary anaerobism due to soil flooding, adequate detention time due to moderate infiltration rate, and a near neutral pH. The infiltration rate at Hollister was in the range reported by Lance and Whisler for good nitrogen removal [79].

Concentrations of all nitrogen species in the shallow water table were statistically indistinguishable (t-test, $\alpha = 0.10$) from the corresponding nitrogen levels in the offsite control wells.

Higher concentrations of nitrate-N and total-N were observed in the intermediate wells and were comparable to offsite control wells. This suggests that the nitrogen in the intermediate aquifer was derived from a source other than the surface applied wastewater.

PHOSPHORUS

Concentrations of phosphorus in municipal wastewaters can range from 1 to 40 mg/L, occurring mostly as inorganic phosphorus (PO_4^{-3}) [84, 85]. The effluent concentration will be a function of influent concentration and the degree of removal during treatment. Primary and secondary effluents may have similar concentrations. For example, the secondary effluent applied at Flushing Meadows ranged from 10 to 15 mg/L PO_4-P [69] while the inorganic phosphorus concentration for Hollister's primary effluent averaged 10.5 mg/L PO_4-P.

Phosphorus removal in rapid infiltration systems has been studied in the Flushing Meadows project [23, 69, 76]. The phosphorus concentration in the wastewater averaged 15 mg/L in 1969, but decreased to about 10 mg/L for the period 1970 to 1972. Phosphorus removal increased with an increase in travel distance and residence time in the soil profile. A travel distance of 9 m (30 ft) removed about 70% of the phosphorus in 1969, but removal efficiency was reduced to about 30% in 1970 when a substantial increase in flowrate occurred. With a flow distance of 100 m (330 ft), phosphorus removal increased to about 90% and was greater with an even longer travel distance. After 5 years of operation and phosphorus additions of nearly 48,000 kg/ha (43,000 lb/acre) the removal efficiency was rather stable.

Pratt [23] points out that rapid infiltration systems require sandy soils that can sustain high water intake rates and high transmissivity in the subsurface environment. Therefore, no layers with high sorptive capacity for phosphorus are likely to be encountered. What little capacity there is may soon be saturated, and the retention will then depend on mineralization and precipitation reactions. One logical precipitant is the calcium supply in the wastewater.

The coarse gravelly soil at Flushing Meadows [69, 76] is calcareous and contains little or no iron and aluminum oxides. Therefore, it was concluded that P removal resulted from the precipitation of calcium phosphate species, ammonium magnesium phosphates, and other insoluble compounds [69].

At the Calumet, Michigan, rapid infiltration system, phosphorus removal continues to be from 89 to 97% efficient, though the system has operated for over 88 years. The influent concentrations of a raw municipal wastewater were reduced from 3.5 mg/L total-P to 0.4 to 0.1 mg/L after 3.1 to 9.2 m (10 to 30 ft) of vertical percolation, respectively. The soil is a poorly sorted gravel having an annual hydraulic loading of 34 m/yr (112 ft/yr) [86].

At Fort Devens, Massachusetts, [73] where a primary effluent has been applied to rapid infiltration basins for over 30 years, removal efficiencies of 86 to 90% were observed in observation wells located 80 to 150 m (262 to 492 ft) from the infiltration basins. Only 18% of the input phosporus remained after 1.5 m (5 ft) of vertical percolation [73]. The total phosphorus level of the effluent was 11 mg/L. The hydraulic loading was 27.1 m/yr (89 ft/yr).

At the Lake George Village rapid infiltration site, total phosphorus in an observation well 4.9 m (16 ft) below the surface averaged 0.9 mg/L, as compared to an input level of 2.1 mg/L. Unchlorinated secondary effluent is applied to natural delta sand beds at an annual hydraulic application rate of 43 m (140 ft).

At Hollister, 85% of the influent phosphorus is inorganic orthophosphate (10.5 mg/L). After percolation through 6.7 m (22 ft) of soil to the shallow groundwater, 23 to 35% of the input P was removed. This figure is in close agreement with the 30% removal ratio calculated for the upper 300 cm of soil profile.

There is no statistical evidence of phosphorus contamination in the intermediate and deep water-bearing strata. Intermediate, deep, and offsite control wells indicate that average P concentrations were statistically indistinguishable at the $\alpha = 0.1$ level. Only the shallow water table responded to wastewater addition.

The ratio of PO_4-P/total P in the wastewater is comparable to the same ratio in the shallow water table. This suggests that there was not selective removal of inorganic-P relative to organic-P in the soil profile. Both forms were entering the shallow groundwater.

FECAL COLIFORM AND TOTAL COLIFORM BACTERIA

The multiple tube, fermentation technique [9] was used to determine the extent of fecal pollution in the unchlorinated primary effluent, renovated water, and groundwater. The presence of fecal coliforms is indicative of pollution from human feces and suggests the presence of pathogenic bacteria. They are an indicator organism that monitors the potential health risks associated with domestic wastewater.

The relative health risks from pathogens at rapid infiltration sites may be greater than other land treatment techniques because the soil must be highly transmissive and thus has less chance to retain bacteria. Bouwer and Chaney [68] point out that fecal coliform bacteria are generally removed after water has traveled 5 to 7 cm (2 to 3 in.) through the soil. However, the coarse soils at rapid infiltration systems, which are necessary to maintain infiltration and transmissivity, may require on the order of 30 m (100 ft) to completely remove fecal coliform bacteria. With adequate travel distance, effective removal of fecal coliforms has been demonstrated [69, 73].

Bacterial removal efficiency within the soil profile is a function of wastewater application rate, subsurface travel distance, solids buildup on the soil surface, the type of organism, soil type, moisture-retention capacity, soil organic content, pH, temperature, sunlight, rain, degree of wastewater contamination, and antagonism from the resident microbial flora. Under certain favorable conditions, applied organisms may actually multiply and increase in numbers. In general, land treatment using intermittent application and drying periods results in die-off of enteric bacteria retained in the soil [23].

Bouwer et al. [69] found deeper penetration of fecal coliforms below rapid-infiltration basins after the basins were flooded following an extended drying or resting period. This was probably due to reduced entrapment of E. coli on the surface of the soil. The clogging layer of organic fines that had accumulated on the soil during flooding, forming an effective filter, partially decomposed after drying. A more open surface was created, resulting in a less effective filter when flooding was resumed. In addition, the bacterial population in the soil undoubtedly declined during drying because the nutrient supply was discontinued. Consequently, there was less competition from the native soil bacteria, and hence greater survival of the fecal coliforms when flooding was resumed. As flooding continued, however, fine suspended solids accumulated again on the soil surface. The native soil bacterial population also increased. This resulted in increased retention of E. coli and a return to fecal coliform levels of essentially zero in renovated water sampled from a depth of 9 m (29 ft). Almost all fecal coliform removal took place in the first 1 m (3 ft) of soil.

At the Fort Devens, Massachusetts rapid infiltration system, analysis of groundwater samples for fecal coliform bacteria proved negative in observation wells located 60 to 100 m (197 to 328 ft) from the application area [73]. Total coliform densities in the unchlorinated effluent ranged from 18 to 53 x 10^6 per 100 mL.

At Hollister, fecal coliform removal approached 99% consistently in the shallow groundwater Wells 5A and 3A after a total percolation distance of 7 m (22 ft). Although fecal coliform densities were substantially reduced from levels in the applied effluent, they were significantly higher than levels in the intermediate and deep wells. In general, fecal coliforms were absent in intermediate and deep wells. The exception was intermediate depth Well 3B which averaged 11 per 100 mL for two samples, with three negative samples. This could have resulted from "short-circuiting" into the intermediate aquifer from either the shallow water table or direct surface contamination. No fecal coliforms were detected in intermediate Well 1B. Total coliform bacteria, which includes fecal coliforms plus coliforms orginating from other sources, were also monitored.

The relationship between fecal and total coliform counts gave some indication of bacteria originating from other than human sources. Total coliform bacteria removal ranged from 96% in Well 5A to 99+% in Well 3A after 7 m (22 ft) of percolation. The uncharacteristically high total coliform densities in Well 2C are believed to be the result of residuals from the drilling operation which were not completely flushed from the well during construction. This material was present in each sample withdrawn from Well 2C over the sampling period, but was not present in samples from other wells. High fecal coliform counts in Well 9C, an offsite control well, were noticed in one sample out of two taken during the study period.

The large heterogeneity of variances in the coliform data made standard statistical comparisons meaningless.

pH

The pH of wastewater can be modified by the soil's buffering capacity and by the chemical and biological reactions which take place within the soil profile at a rapid infiltration site. Wastewater which contains organic acids can show an increase in pH as it moves through the soil because the acids are biodegraded and lost from the system [68]. On the other hand, for biodegradable material, pH may decrease because soil microbial activity produces CO_2 and organic acids [68].

At the Flushing Meadows project, the pH of sewage effluent was about 8, whereas that of the renovated water was approximately 7. This pH drop was attributed to the bacterial production of CO_2 and organic acids. Nitrification and removal of carbonates would also reduce the pH [69].

At the Lake George Village, New York, rapid infiltration site, secondary effluent pH was 6.94 while that of the renovated water in a well 4.9 m (16 ft) directly beneath an infiltration basin was slightly lower at 6.37 [87]. Production of organic acids and carbon dioxide by soil bacteria and nitrification of ammonium were thought to be responsible for the slight pH drop [88].

At Fort Devens, Massachusetts, the pH of soil water extracted by suction lysimeters at 1.5 m (5 ft) increased slightly in one lysimeter and stayed the same in another [73]. Control site soil pH ranged from 4.6 to 5.2 whereas the treatment bed pH ranged from 5.7 to 6.2 [73]. Comparison of effluent and groundwater pH directly beneath spreading basins at a depth of 24.4 m (80.1 ft) showed a drop from 7.0 to 6.8 standard units [73].

At the Whittier Narrows and Rio Hondo test basins, surface applied secondary effluent had pH values of 8.0 and 8.30, respectively. After percolation through 1.8 m (6 ft) of soil previously used for groundwater recharge of stormwater runoff, the pH had dropped to 7.78 and 7.90, respectively. The pH reduction was attributed to carbon-dioxide production and/or nitrification in the soil matrix [89].

At Hollister, a significant difference between the average pH of the primary effluent (7.3) and the pH of the shallow groundwater (7.6) was observed. However, the pH of the shallow groundwater was indistinguishable (t-test, $\alpha = 0.10$) from that of the offsite control wells (7.7). This suggests that wastewater dissolution of soil calcium carbonate is increasing the pH of the percolating solution, offsetting the production of hydrogen ions from nitrification reactions and organic acid formation.

DISSOLVED SOLIDS

Total Dissolved Solids and Electrical Conductivity

The total dissolved solids (TDS) concentration of municipal wastewater is comprised of several ions including sodium (Na^+), calcium (Ca^{+2}), magnesium (Mg^{+2}), potassium (K^+), chloride (Cl^-), sulfate (SO_4^{-2}), and others. Effluent TDS of several primary treatment plants in California range from 935 to 1,898 mg/L [84]. The secondary effluent at Flushing Meadows ranged from 1,000 to 1,200 mg/L [69].

TDS concentrations can vary widely depending on the industrial mix of the community, the proportion of commercial to residential development, and the nature of the residential community. Wide variations can be encountered from drainage areas having selected land development characteristics yet having the same water supply [84].

The effluent and shallow groundwater TDS concentrations at Hollister were both generally in the 1,200 to 1,300 mg/L range. The salt concentration of the renovated water is approximately equal to that of the applied effluent. Electrical conductivity (EC) measurements bear this out. TDS and EC measurements in the intermediate and deep observation wells were highly variable. Constrasts using the t-statistic revealed that TDS and EC measurements in the effluent, shallow groundwater, and offsite control wells were generally indistinguishable.

Exchangeable Cations and Sodium Adsorption Ratio

Though all four of the major cations (Na^+, K^+, Ca^{+2}, Mg^{+2}) found in wastewater undergo exchange reactions with soils, the fate of applied Na^+ is generally of greatest concern. When the wastewater ratio of sodium to calcium and magnesium is large, highly hydrated sodium cations replace less hydrated Ca and Mg on the soil. This process causes the dispersion of soil clay particles, resulting in a decrease in soil permeability. The nature of rapid infiltration systems is such that the sodium hazard is generally considered minimal, because the clay percentage is usually so small that the effect is not noticed.

If the reuse of renovated water for irrigation purposes is an objective, sodium hazard must be considered. In most cases soil permeability becomes a hazard before direct sodium toxicity is noticed. In a few plants this is not strictly true, notably avocados [52]. To determine the sodium hazard, the sodium adsorption ratio (SAR) was developed by the U.S. Department of Agriculture Salinity Laboratory. SAR is defined as follows:

$$SAR = Na/[1/2\ (Ca + Mg)]^{1/2}$$

where Na, Ca, and Mg are concentrations of the respective ions in meq/L of water. Ratios less than 4 are generally acceptable on almost all soils, though higher values are suitable for water of higher conductivity. A detailed description of the SAR and sodium hazard is found in reference [12].

At Hollister, concentrations of sodium and potassium in the shallow aquifer were not statistically different ($\alpha = 0.10$) from their concentrations in the wastewater, but were greater than control well concentrations. This suggests that the soil's exchange capacity for those cations has been exhausted. Increases in exchangeable sodium and potassium were noted throughout the soil profile, and are consistent with this premise. It appears that Na and K were entering the shallow groundwater unimpeded.

In contrast to Na and K, shallow groundwater concentrations of Ca and Mg were greater than input levels, but were still less than or equal to control well values. As discussed previously, these increases may be related to wastewater solubilization of $CaCO_3$, resulting in increased Ca input to the shallow aquifer. The resultant Ca-Mg exchange reactions would also tend to increase Mg concentrations of the percolating effluent.

Although SAR values in the shallow aquifer were 40% higher than in the offsite control wells, the value is considerably lower than the primary effluent value of 5.71. Because sodium levels have remained constant, both Ca^{+2} and Mg^{+2} must necessarily account for the decrease in SAR.

Background conductivity values at Hollister ranged upwards of 2,000 μmhos/cm. Therefore, conductivity, as opposed to sodium, poses the biggest threat to potential reuse of shallow groundwater for irrigation purposes. According to the USDA [12], the shallow groundwater is classified as highly saline-low sodium water. It is suitable for use on almost all soils with little danger of the development of harmful levels of exchangeable sodium, as long as restricted drainage is not a problem.

MAJOR ANIONS

Chloride and sulfate are the major anions in both the wastewater and renovated water at the Hollister rapid infiltration site. The significance of chloride is its ion specific toxicity, when applied to sensitive crops. Above 4 meq/L, chloride becomes an increasing problem and above 10 meq/L it becomes a severe problem to sensitive crops [52]. The chloride level of the effluent and renovated water (as measured by Wells 3A and 5A) averaged between 280 and 291 mg/L or 7.9 to 8.2 meq/L. At this level some chloride toxicity would be expected if the water were to be used on sensitive plants, such as peaches or other stone fruits [52]. However, it should be noted that shallow groundwater chloride levels were no different than concentrations in the offsite control wells.

Sulfate ion is one of the major anions occurring in natural waters [90]. It is of importance in public water supplies because of its cathartic effect upon humans when it is present in excessive amounts [66]. The U.S. Public Health Service recommends an upper limit of 250 mg/L in waters intended for human consumption. Sulfates are also important, primarily in industrial reuse applications, because of their tendency to form scales in boilers and heat exchangers.

The sulfate concentration in the effluent and shallow groundwater at Hollister was less than that contained in the majority of potable water wells serving the City of Hollister. The sulfate concentration of Well 3A was noticeably less than the effluent, while Well 5A was somewhat higher than the effluent sulfate measured. An analysis-of-variance reveals that the sulfate level in the effluent, treatment site aquifers, and offsite control wells are indistinguishable ($p = 0.78$). Sulfate contamination resulting from wastewater application has not occurred.

SUSPENDED SOLIDS

The suspended solids (SS) concentration of untreated municipal wastewater can range from 100 to 350 mg/L [85]. Municipal effluent from several primary treatment plants indicates that the suspended solids are completely removed in most cases [85], if the facility is not overloaded. The SS in the primary effluent at Hollister averages 274 mg/L, indicating the lack of treatment efficiency by the pretreatment facility.

The average removal indicated by total SS levels in Wells 3A and 5A was 95%. Effective filtering of input solids occurred although complete removal was not indicated after percolation through 7.7 m (22 ft) of soil. Improper well development is probably the cause of inorganic SS levels, which were also detected in several intermediate and deep observation wells.

Complete evaluation of removal effectiveness cannot be made because SS levels in the offsite control wells were not measured. However, the general appearance of these samples indicated that no SS were present.

ALKALINITY

Bicarbonate accounted for all of the alkalinity in the Hollister public water supply (see Table 4) with additional pickup during domestic water use. Biological activity, during wastewater percolation through the soil profile, can produce carbon dioxide and organic acids. The resulting action of CO_2 solubilizes soil carbonate materials and can cause an additional increase in the alkalinity of the percolate.

Results at Lake George Village, New York, showed seasonal increases in alkalinity in shallow observation wells during the summer months [88]. The well was located 5 m (16 ft) below the surface. Natural dilution and biological slowdown are given as reasons for decreases in alkalinity during winter months [88].

Alkalinity in the primary effluent at Hollister averaged 446 mg/L, which is roughly 110 mg/L greater than the domestic water supply. In the shallow observation wells, alkalinity increases were observed relative to input levels, though the increases were not significant. The simultaneous reduction in BOD during percolation implies considerable biological activity and CO_2 production. The resulting attack on soil carbonate material would tend to increase alkalinity.

Alkalinity measurements from the intermediate and deep observation wells do not indicate any influence from application of wastewater to the soil system. In fact, all measurements are statistically indistinguishable.

BORON

The primary environmental concern with boron in renovated water is its potential toxicity to plants. Accumulation of excessive boron in plant tissue results in reduced production. Toxicity limits for boron in water applied to sensitive crops are <0.75 mg/L--no problem; 0.75 to 2.0 mg/L--increasing problem; and >2.0--severe problem [12]. Boron levels in some surface waters used for irrigation in western United States range from 0.03 to 0.46 mg/L. Borates in household detergents can increase the boron concentration in domestic wastewater significantly. Bouwer et al. [69] found that boron in the secondary effluent from Phoenix, Arizona, had risen from 0.4 mg/L in 1969 to 0.9 mg/L in 1971.

The boron concentration of the Hollister public water supply ranges from 0.05 to 0.7 mg/L for four main supply wells (See Table 4). As a result of mineral pickup during domestic and commercial use, wastewater boron concentrations are twice that of the raw water supply, averaging 1.4 mg/L.

Very few results on boron removal at rapid infiltration sites have been reported. Bouwer et al.[69] found essentially no boron removal in the sandy and gravelly soils below infiltration basins at Flushing Meadows. The lack of removal was attributed to absence of iron and aluminum oxides in the soil, which have the ability to fix boron.

At Hollister very little, if any, boron is currently removed after percolation through 7 m (22 ft) of the soil profile. Based on average concentrations, only 14% of the applied boron is retained in the overlying soil. The absence of a significant amount of clay is believed to be the reason for low boron removal.

Deep aquifer wells have boron concentrations that are significantly lower than offsite control, shallow, or intermediate wells. The boron concentrations in offsite control wells, shallow and intermediate wells are not significantly different.

FLUORIDE

Fluoride is significant in public drinking water supplies due to its well recognized effects on human teeth [90]. Evidence from public health studies has shown that fluoride-ion concentrations of approximately 1.0 mg/L are optimum for the prevention of dental fluorosis and cavities [90].

Fluoride is also significant in irrigation water. A maximum limit of 2.0 mg/L has been set by the National Academy of Science and National Academy of Engineering for continuous use on all soils [66]. Applications of soluble fluoride salts to acid soils can produce toxicity to plants [91]. Soil type, calcium and phosphorus content, and pH seem to be the major factors controlling plant uptake of fluoride [92]. The adsorption of wastewater fluoride by soil and its subsequent equilibration with fluorite (CaF_2) and fluorapatite ($Ca_5 [PO_4]_3 F$) leads to both retention of fluoride in the soil and the control of ion-specific injury to plants. The Ca^{+2} added with wastewater generally maintains the soil Ca level high enough to prevent fluoride injury. Injury from added NaF has been demonstrated in acidic soils low in Ca, but not in well-limed soils [93]. Crops raised on fluoride-enriched soils show little increased fluoride uptake as long as the soil is near neutral pH.

Very little information on the fate of fluoride at operating rapid infiltration systems is available. At Flushing Meadows, Arizona, the fluoride content of secondary effluent was reduced from 4.1 to 2.6 mg/L after movement through 9 m (30 ft) of sandy soil in a high rate application system. Increased removal was noted with further movement through the coarse textured soil [69]. The fluoride removal somewhat paralleled the phosphate removal suggesting precipitation of fluorapatite.

At Hollister, apparently no fluoride is added to the municipal wastewater from domestic or other water use since concentrations in potable water and primary effluent are relatively equal. The fluoride concentrations in Wells 3A and 5A increased after 7 m (22 ft) of movement through the soil profile. This suggests that F^- is currently passing through the soil profile without treatment, causing concentration levels to rise relative to baseline levels. It may also be possible that the decrease in soil pH resulting from wastewater addition may cause the dissolution of any fluorapatite species present in the soil profile.

It should be noted that even the average fluoride concentration of 1.0 mg/L in Well 3A is below the irrigation water limit set for use on all soils [66].

TRACE ELEMENTS

The greatest concern regarding trace elements has been their potential health and plant toxicity hazard [68]. Soil has been shown to be effective in reducing the concentration of trace elements in percolating effluent over limited periods of time. However, their long term ability to remove metals is questioned because of their ineffectiveness after sorption saturation [60]. This concern would be especially true if the creation of new soil sorption sites (influx of wastewater clays and organic material) did not keep pace with the influx of metals.

Regulations regarding the recommended maximum concentrations of trace elements in irrigation waters and maximum allowable concentrations of trace elements in drinking water have been established. These are tabulated in Table 30, together with the average effluent concentrations of the Hollister wastewater. Except for lead, all average effluent concentrations of trace elements are below both of these limits. Effluent lead slightly exceeds drinking water standards.

TABLE 30. COMPARISON OF TRACE ELEMENT LEVELS TO IRRIGATION AND DRINKING WATER LIMITS
mg/L

Element	Recommended maximum in irrigation waters [66]	Maximum concentration in drinking waters [94]	Average wastewater concentration	Average shallow groundwater concentration
Ag (silver)	--[a]	0.05	<0.008	<0.006
As (arsenic)	0.1	0.05	<0.01	<0.01
Ba (barium)	--[a]	1.0	<0.13	<0.13
Cd (cadmium)	0.01	0.010	<0.004	0.028
Co (cobalt)	0.1	--[a]	<0.008	0.010
Cr (chromium)	0.05	0.05	<0.014	<0.014
Cu (copper)	0.2	--[a]	0.034	0.038
Fe (iron)	5.0	--[a]	0.39	0.36
Hg (mercury)	--[a]	0.002	<0.001	<0.001
Mn (manganese)	0.2	--[a]	0.070	0.96
Ni (nickel)	0.2	--[a]	0.051	0.13
Pb (lead)	5.0	0.05	0.054	0.09
Se (selenium)	0.02	0.01	<0.001	<0.001
Zn (zinc)	2.0	--[a]	0.048	0.081

[a] None set.

Most studies investigating solution heavy metal-solid phase interactions have found that specific adsorption, interfacial precipitation, bulk precipitation, and cation exchange can reduce influent heavy metal concentrations. Mobility potentials of heavy metals are increased in acidic soils, having low ion-exchange capacities and organic matter.

Several studies have reported on the fate of trace metals at operating rapid infiltration projects. Metal concentrations in the secondary effluent and in the renovated water from a well 27 m (86 ft) from the basins of the Flushing Meadows project showed considerable removal of copper and zinc, but not of cadmium and lead [69]. Metals did not accumulate in the surface 1.5 m (5 ft) of the soil because of the low organic matter and clay content of the basin soils, the low retention times of the water in the surface soils, and the low metal concentrations in the effluent.

A study of metal accumulation in infiltration basins at Fort Devens, Massachusetts, revealed a peak of heavy metals which coincided with an organic matter accumulation zone at 45 cm (18 in.). The organic matter in this zone and its metal content appeared to increase during winter and decrease during summer [95].

The soils at Hollister are low in organic matter with a cation exchange capacity typical of sandy and gravelly soils. The soil pH is generally alkaline favoring heavy metal sorption.

Of the 14 trace elements monitored in the Hollister groundwater, five showed no significant increases in groundwater concentrations relative to offsite control wells; three had concentrations that averaged below the limits of detection; and six showed significant increases in at least one aquifer relative to background levels in the control wells.

Groundwater concentrations of Ag, Ba, Cd, Co, and Cr in all three aquifers below the treatment site have been unaffected by wastewater addition. No statistical differences were observed between treatment site aquifers and offsite control wells. Concentrations, of Ag, Ba, Cd, Co, and Cr in the offsite control wells are not significantly different from concentrations in the primary effluent.

The F ratio for Cd is nearly significant ($p = 0.11$). This suggests that the shallow aquifer may be exhibiting the first signs of Cd "breakthrough." The average concentration of Cd in the shallow wells exceeds maximum allowable concentrations in drinking water, but because of its variability, is not significantly different from the offsite control site wells. There is no soil evidence that suggests Cd mobilization through the upper 300 cm of soil (see Figure 41).

Analyses-of-variance could not be performed for As, Hg, and Se because all reported values were below the limit of flame atomic absorption detection. While no conclusive statement regarding the transport and removal of these elements through the soil profile can be made, it is apparent that groundwater contamination by these metals is not occurring.

Treatment site aquifer concentrations of Mn, Ni, Fe, Zn, Pb, and Cu were significantly different than control well values. Significant shallow groundwater-control well contrasts were observed for all six metals. Average control well values of 0.006, 0.041, 0.003, 0.021, 0.016 and 0.015 mg/L were observed for Mn, Ni, Fe, Zn, Pb, and Cu, respectively. This compares with average treatment site shallow groundwater values of 0.961, 0.125, 0.356, 0.081, 0.074, and 0.038 mg/L for the same metals. Intermediate well-control well contrasts were observed for all of these metals except Ni. Deep well-control well contrasts were significant for Cu, Mg, Pb, and Zn.

Average shallow and intermediate groundwater concentrations of Pb exceeded those limits set by the EPA for drinking water. The concentration of Mn in the same groundwaters exceeded the recommended maximum concentration for irrigation waters used continuously on all soils.

Metal concentrations in some wells were greater than that input by wastewater. This is particularly apparent for Mn. Input Mn concentrations averaged 0.070 mg/L, as compared to average values of 0.961, 0.357, 0.154 mg/L for shallow, intermediate, and deep groundwater, respectively. Lehman and Wilson [60] observed similar behavior in a continuously flooded, lysimeter-equipped soil column. In their study, more solution Mn was observed in soil-water at depths of 15 and 23 cm (6 and 9 in.), than at 8 cm (3 in.). They concluded that Mn originated from within the soil profile. In addition, Ng and Bloomfield [61, 96] studied trace metal mobilization in soil media and found that Mn was mobilized by continuous flooding in the presence of organic matter. Mn mobilization can occur if pH-Eh conditions are modified to favor the conversions of tetravalent, insoluble manganic forms to divalent, soluble manganous species. Hem [35] has shown that the solubility of Mn^{+2} is enhanced when both pH and Eh are lowered. The solubility of iron also increases when conditions become more acidic and more anaerobic zones are created.

At Hollister, a significant decrease in soil pH was observed after wastewater addition. In addition, the input of an organic rich wastewater can create anaerobic environments in the soil. Immediately after flooding, the soil surface becomes saturated; oxygen transfer is diminished; and a decrease in redox potential results. Both the lowered pH and Eh conditions favor the solubility of soil Mn and Fe. This is substantiated by the significant loss of soil Mn from the upper 30 cm of soil. Significant deposition was not detected through 300 cm, suggesting that Mn is travelling to the underlying groundwater. In contrast, soil Fe showed significant increases throughout the soil profile. However, the reduction of ferrous iron to ferric, followed by re-oxidation (after drying) to ferric and subsequent precipitation as a hydrous oxide, could lead to a less crystalline oxide, more amenable to DTPA extraction. Therefore, mobilization of soil Fe cannot be ruled out.

In addition to the mobilization of Mn and Fe, the possibility exists that the wastewater addition has resulted in the mobilization of soil Ni, Zn, Pb, and Cu. Ng and Bloomfield [61, 96] noted that these metals were mobilized by continuous flooding in the presence of organic matter. While the formation of metal chelates may be an important factor in the transport of these metals, the more likely possibility is that mobilization has occurred in response to the dissolution of iron and manganese oxides. It has often been suggested that these oxides furnish the principal control on the fixation of heavy metals in soils. Consequently, their loss from the soil profile could decrease the ability of the soil to retain input heavy metals, and dissolve existing sorbed metals.

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TABLE A-1. WASTEWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Parameter	6/21/76	6/23/76	6/24/76	9/15/76	9/16/76	9/17/76	12/6/76	12/7/77	12/8/76	2/22/76	2/23/77	2/24/77
COD	680	546	647	572	625	581	656	722	598	840	969	1,029
BOD	150	151	219	211	214	202	134	145	180	414	359	266
TOC	--	--	--	--	--	--	--	--	--	240	240	264
Total Kjeldahl Nitrogen	46.2	38.9	39.5	35.3	58.3	45.6	30.8	29.2	31.2	40.2	40.2	41.6
NH_3-N	24.4	22.4	19.7	23.0	44.0	33.6	21.7	22.5	22.8	22.6	23.9	23.0
N-Organic	21.8	16.50	19.8	12.3	14.3	12.0	9.1	6.7	8.4	17.6	16.3	18.6
NO_3-N	0.68	0.64	0.76	0.17	0.19	0.67	0.46	0.44	0.52	0.2	0.16	0.18
Total P	19.5	22.0	24.0	10	12	10	21.5	11.3	15.9	10.0	10.0	10.5
PO_4-P	13.2	12.8	12.0	9	9	9	8.5	11.3	12.2	9.2	9.9	10.4
Total Coliforms, count/100 ml	1.3×10^6	5.4×10^6	49,000	46×10^6	4.6×10^6	4.6×10^6	92×10^6	92×10^6	24×10^6	16×10^6	24×10^6	24×10^6
Fecal Coliforms, count/100 ml	--	--	--	11×10^6	4.6×10^6	4.6×10^6	9.2×10^6	2.2×10^6	3.5×10^6	16×10^6	16×10^6	24×10^6
pH, units	7.0	7.0	7.0	7.2	7.3	7.3	7.5	7.8	8.1	7.3	7.1	7.1
TDS	1,256	1,184	1,152	1,500	1,593	1,585	1,032	1,042	1,024	1,072	1,044	1,016
Conductivity, µmhos/cm	1,890	1,790	1,860	2,320	2,400	2,480	1,600	1,640	1,620	1,230	1,330	1,320
SS	--	--	--	--	--	--	234	222	206	328	327	327
Alkalinity	--	--	--	--	--	--	447	437	416	454	465	459
B	1.3	1.0	1.1	1.8	1.8	1.7	1.2	1.6	1.4	1.4	1.3	1.3
F	0.79	0.50	0.45	0.55	0.6	0.55	0.37	0.41	0.33	1.1	1.1	1.1
Na	253	244	242	363	377	391	230	230	230	196	196	200
Ca	56	64	58	70	71	71	55	52	53	32	31	31
Mg	60	61	64	76	69	27	74	75	75	62	62	64
K	11.7	11.3	10.9	11.7	12.5	12.1	12.5	12.9	12.5	16.0	14.8	16.0
Cl	260	266	248	434	488	490	185	200	190	220	205	224
SO_4	161	180	186	218	228	210	250	228	250	225	210	210
Ag	0.008	0.008	0.006	0.008	< 0.005	< 0.005	0.010	0.011	0.012	0.011	<0.005	<0.005
As	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Ba	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.11	0.10	0.12	0.24	0.22	0.18
Cd	0.004	0.005	0.004	<0.001	<0.001	<0.001	0.007	0.007	0.008	0.003	0.003	0.002
Co	0.008	0.008	0.007	0.014	0.009	0.010	<0.006	<0.006	<0.006	<0.006	<0.006	<0.006
Cr	<0.004	<0.004	<0.004	<0.004	<0.004	<0.004	0.012	0.018	0.015	0.030	0.036	0.031
Cu	0.024	0.028	0.019	0.031	0.021	0.020	0.070	0.070	0.065	0.030	0.032	0.035
Fe	0.34	0.50	0.38	0.15	0.22	0.14	0.35	0.32	0.34	0.52	0.82	0.54
Hg	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Mn	0.056	0.058	0.094	0.070	0.067	0.055	0.067	0.067	0.071	0.080	0.084	0.075
Ni	0.065	0.065	0.060	0.092	0.087	0.080	0.016	0.015	0.015	0.048	0.038	0.033
Pb	0.012	0.025	0.012	0.045	0.033	0.022	0.070	0.068	0.065	0.120	0.110	0.070
Se	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Zn	0.016	0.020	0.010	0.035	0.030	0.030	0.052	0.051	0.057	0.090	0.090	0.080

TABLE B-1. pH
Units

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	8.500	7.800	6.300	6.400	6.700	7.140	13.528
2.	25-35	9.200	8.400	6.800	6.500	7.600	7.700	14.520
3.	95-105	9.400	8.900	7.900	7.800	8.000	8.400	8.460
4.	295-305	9.400	9.100	8.800	8.800	8.600	8.940 8.045	3.502
5.	TOTAL	36.500	34.200	29.800	29.500	30.900	160.900	
6.	MEAN	9.125	8.550	7.450	7.375	7.725		
7.	RSD	4.682	6.786	15.047	84.333	55.285		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	8.150	6.467	-5.545880
25-35	8.800	6.967	-3.537971
95-105	9.150	7.900	-6.228411
295-305	9.250	8.733	-3.661871

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

8.940 8.400
8.400
7.700
7.140

****

9.125 8.550
8.550
7.725 7.450 7.375

---

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-2. CALCIUM CARBONATE
Percent

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	1.550	1.080	0.280	0.360	0.000	0.654	97.857
2.	25-35	2.270	2.060	0.270	0.290	0.000	0.978	111.671
3.	95-105	2.470	2.270	1.170	2.010	0.000	1.584	64.052
4.	295-305	2.510	3.040	1.420	1.390	2.320	2.136	33.598
							1.338	
5.	TOTAL	8.800	8.450	3.140	4.050	2.320	26.760	
6.	MEAN	2.200	2.113	0.785	1.013	0.580		
7.	RSD	20.267	38.202	76.139	227.059	427.675		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	1.315	0.213	-4.900790
25-35	2.165	0.187	-13.750675
95-105	2.370	1.060	-1.732522
295-305	2.775	1.710	-2.416834

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

2.136	1.534	
1.534	0.978	
0.978	0.654	

****

2.200	2.113	
2.113		
1.013	0.785	0.580

---

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-3. ELECTRICAL CONDUCTIVITY SOIL EXTRACT
mmhos/cm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.740	0.770	2.943	2.623	2.467	1.909	55.906
2.	25-35	0.100	0.500	2.400	1.280	0.770	1.010	87.874
3.	95-105	0.100	0.330	1.800	1.260	0.500	0.798	88.886
4.	295-305	0.400	0.500	0.700	0.830	0.600	0.616	30.057
							1.083	
5.	TOTAL	1.340	2.100	7.843	6.043	4.337	21.663	
6.	MEAN	0.335	0.525	1.961	1.511	1.084		
7.	RSD	90.984	34.654	49.034	110.773	99.628		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	0.755	2.678	10.609833
25-35	0.300	1.483	1.851530
95-105	0.215	1.187	1.965833
295-305	0.450	0.727	2.467282

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

1.909		
1.010	0.798	0.616

****

1.961	1.511
1.511	1.084
1.084	0.525
0.525	0.335

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[a] Relative standard deviation

[b] Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-4. ORGANIC MATTER
Percent

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.760	1.350	1.200	2.190	2.020	1.504	39.427
2.	25-35	0.080	0.150	0.200	0.120	0.280	0.166	46.585
3.	95-105	0.020	0.020	0.060	0.030	0.050	0.036	50.461
4.	295-305	0.010	0.020	0.110	0.080	0.030	0.050	86.023
							0.439	
5.	TOTAL	0.870	1.540	1.570	2.420	2.380	8.780	
6.	MEAN	0.218	0.385	0.393	0.605	0.595		
7.	RSD	166.890	167.856	137.947	89.501	63.607		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	1.055	1.803	1.656706
25-35	0.115	0.200	1.305976
95-105	0.020	0.047	2.342160
295-305	0.015	0.073	1.921840

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

1.504
0.166   0.050   0.036

****

0.605   0.595   0.393   0.385   0.218

---

a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-5. TOTAL NITROGEN
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	559.000	840.000	1182.000	1722.000	1586.000	1177.800	41.590
2.	25-35	76.000	112.000	372.000	347.000	200.000	221.400	60.608
3.	95-105	64.000	34.000	117.000	93.000	104.000	82.400	40.511
4.	295-305	45.000	184.000	178.000	187.000	77.000	134.200	50.559
							403.950	
5.	TOTAL	744.000	1170.000	1849.000	2349.000	1967.000	8079.000	
6.	MEAN	186.000	292.500	462.250	587.250	491.750		
7.	RSD	133.868	126.531	106.435	100.985	54.328		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	699.500	1496.667	3.405576
25-35	94.000	306.333	3.009486
95-105	49.000	104.667	3.886174
295-305	114.500	147.333	0.476101

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

1177.800			
221.400	134.200	82.400	

****

587.250	491.750	462.250	292.500
491.750	462.250	292.500	186.000

a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-6. ORGANIC NITROGEN
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	545.000	818.000	1122.000	1592.000	1491.000	1113.600	39.706
2.	25-35	62.000	98.000	326.000	290.000	174.000	190.000	60.925
3.	95-105	45.000	30.000	99.000	74.000	86.000	66.800	42.912
4.	295-305	34.000	173.000	137.000	150.000	65.000	111.800	53.073
							370.550	
5.	TOTAL	686.000	1119.000	1684.000	2106.000	1816.000	7411.000	
6.	MEAN	171.500	279.750	421.000	526.500	454.000		
7.	RSD	145.345	129.957	113.482	98.373	43.108		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	681.500	1401.667	3.419416
25-35	80.000	263.333	3.020034
95-105	37.500	86.333	4.493534
295-305	103.500	117.333	0.222999

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

```
1113.600
 190.000     111.800     66.800

    ********

 526.500     454.000     421.000     279.750
 454.000     421.000     279.750     171.500
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a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-7. TOTAL PHOSPHORUS
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	705.000	792.000	1560.000	1965.000	2820.000	1568.400	55.877
2.	25-35	609.000	456.000	1192.000	1440.000	1403.000	1020.000	44.919
3.	95-105	385.000	363.000	719.000	788.000	923.000	635.600	39.324
4.	295-305	334.000	468.000	964.000	842.000	850.000	691.600	39.584
							978.900	
5.	TOTAL	2033.000	2079.000	4435.000	5035.000	5996.000	19578.000	
6.	MEAN	508.250	519.750	1108.750	1258.750	1499.000		
7.	RSD	34.905	36.069	32.243	104.694	107.582		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	748.500	2115.000	2.843629
25-35	532.500	1345.000	7.073207
95-105	374.000	810.000	5.605943
295-305	401.000	885.333	6.794494

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

1568.400		
1020.000	691.600	635.600

****

1499.000	1258.750	1108.750
1258.750	1108.750	
1108.750		
519.750	508.250	

---

a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-8. BICARBONATE EXTRACTABLE PHOSPHORUS, ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	13.000	8.400	57.000	109.000	111.000	59.680	83.285
2.	25-35	0.670	0.150	61.000	73.000	63.000	39.564	91.070
3.	95-105	0.560	0.090	42.000	39.000	33.000	22.930	91.099
4.	295-305	3.700	0.170	54.000	32.000	21.000	22.174	99.223
							36.087	
5.	TOTAL	17.930	8.810	214.000	253.000	228.000	721.740	
6.	MEAN	4.483	2.203	53.500	63.250	57.000		
7.	RSD	130.770	187.596	15.300	114.806	115.156		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	10.700	92.333	3.567259
25-35	0.410	65.667	13.606802
95-105	0.325	38.000	11.015636
295-305	1.935	35.667	2.678614

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

59.680	39.564	
39.564	22.930	22.174

****

63.250	57.000	53.500
57.000	53.500	
53.500		
4.483	2.203	

[a] Relative standard deviation
[b] Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-9. BORON
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.190	0.220	0.660	1.080	0.700	0.570	65.173
2.	25-35	0.170	0.150	0.360	0.350	0.340	0.274	38.156
3.	95-105	0.140	0.090	0.180	0.170	0.160	0.148	24.079
4.	295-305	0.130	0.170	0.380	0.470	0.200	0.270	54.496
							0.316	
5.	TOTAL	0.630	0.630	1.580	2.070	1.400	6.310	
6.	MEAN	0.158	0.158	0.395	0.518	0.350		
7.	RSD	17.484	34.142	50.188	109.863	101.967		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	0.205	0.813	3.513570
25-35	0.160	0.350	18.024983
95-105	0.115	0.170	2.740501
295-305	0.150	0.350	1.931468

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

0.570
0.274   0.270   0.143

****

0.518   0.395   0.350
0.395   0.350   0.158   0.158

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-10. CATION EXCHANGE CAPACITY
meq/100 g

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	7.170	8.640	4.820	11.170	10.740	8.508	30.805
2.	25-35	2.790	3.250	4.140	6.210	4.160	4.110	31.945
3.	95-105	1.910	1.240	3.190	2.700	1.980	2.204	34.286
4.	295-305	2.310	2.580	6.930	7.500	3.230	4.510	55.432
							4.833	
5.	TOTAL	14.180	15.710	19.080	27.580	20.110	96.660	
6.	MEAN	3.545	3.928	4.770	6.895	5.028		
7.	RSD	53.025	63.681	25.606	71.152	24.511		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	7.905	8.910	0.372073
25-35	3.020	4.837	2.011952
95-105	1.575	2.623	2.024490
295-305	2.445	5.887	1.988362

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

8.508			
4.510	4.110	2.204	

****

6.895	5.028	4.770	
5.028	4.770	3.928	3.545

a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-11. EXCHANGEABLE SODIUM, meq/100 g

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.930	0.410	1.210	3.410	1.230	1.438	80.039
2.	25-35	0.110	0.070	0.770	0.590	0.480	0.404	75.517
3.	95-105	0.001	0.020	0.390	0.220	0.260	0.178	92.943
4.	295-305	0.120	0.050	0.910	0.740	0.140	0.392	102.350
							0.603	
5.	TOTAL	1.161	0.550	3.280	4.960	2.110	12.061	
6.	MEAN	0.290	0.138	0.820	1.240	0.528		
7.	RSD	148.111	132.964	41.511	106.775	105.954		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	0.670	1.950	1.330321
25-35	0.090	0.613	4.751761
95-105	0.011	0.290	4.195057
295-305	0.085	0.597	1.690705

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

1.438			
0.404	0.392	0.178	

****

1.240	0.820	0.528	
0.820	0.528	0.290	0.138

---

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-12. EXCHANGEABLE POTASSIUM, meq/100 g

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.260	0.300	0.220	0.340	0.380	0.300	21.082
2.	25-35	0.020	0.050	0.130	0.160	0.100	0.092	62.156
3.	95-105	0.001	0.001	0.060	0.010	0.090	0.032	124.926
4.	295-305	0.010	0.001	0.160	0.160	0.020	0.070	117.166
							0.124	
5.	TOTAL	0.291	0.352	0.570	0.670	0.590	2.473	
6.	MEAN	0.073	0.088	0.143	0.168	0.148		
7.	RSD	171.923	162.737	46.681	72.404	28.968		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	0.280	0.313	0.522233
25-35	0.035	0.130	3.800000
95-105	0.001	0.053	1.737310
295-305	0.006	0.113	1.787104

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

0.300			
0.092	0.070	0.032	

****

0.168	0.148	0.143	0.088
0.148	0.143	0.088	0.073

---

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-13. EXCHANGEABLE MAGNESIUM, meq/100 g

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	1.840	1.810	3.300	6.970	6.320	4.048	60.692
2.	25-35	1.120	1.140	2.880	4.020	2.590	2.350	52.568
3.	95-105	1.250	0.700	2.240	1.310	1.430	1.386	39.946
4.	295-305	1.480	7.590	4.180	4.210	1.880	3.868	62.960
							2.913	
5.	TOTAL	5.690	11.240	12.600	16.510	12.220	58.260	
6.	MEAN	1.423	2.810	3.150	4.128	3.055		
7.	RSD	22.189	114.562	25.819	108.126	95.136		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	1.825	5.530	2.538119
25-35	1.130	3.163	3.608478
95-105	0.975	1.660	1.596132
295-305	4.535	3.423	-0.447265

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

4.048 3.868 2.350 1.386

****

4.128 3.150 3.055 2.810 1.423

---

a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-14. EXCHANGEABLE CALCIUM, meq/100 g

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	5.550	5.660	0.640	1.670	0.940	2.892	86.621
2.	25-35	3.020	1.820	0.010	0.090	0.180	1.024	131.297
3.	95-105	1.750	0.880	2.790	2.310	0.570	1.660	56.382
4.	295-305	1.450	3.150	4.510	6.240	2.720	3.614	50.637
							2.298	
5.	TOTAL	11.770	11.510	7.950	10.310	4.410	45.950	
6.	MEAN	2.943	2.878	1.988	2.578	1.103		
7.	RSD	63.442	72.134	103.658	96.381	336.062		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	5.605	1.083	-11.390349
25-35	2.420	0.093	-5.151093
95-105	1.315	1.890	0.618915
295-305	2.300	4.490	1.503232

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

3.614 2.892 1.660 1.024

****

2.943 2.878 2.578 1.988 1.103

---

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-15. DTPA-IRON
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	19.000	23.000	122.000	274.000	169.000	121.400	88.066
2.	25-35	7.560	6.990	104.000	160.000	140.000	83.710	86.732
3.	95-105	6.300	3.700	41.000	53.000	28.000	26.400	81.298
4.	295-305	7.000	7.990	26.000	32.000	14.000	17.398	63.965
							52.227	
5.	TOTAL	39.860	41.680	293.000	519.000	351.000	1244.540	
6.	MEAN	9.965	10.420	73.250	129.750	87.750		
7.	RSD	60.666	32.386	64.008	115.035	114.235		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	21.000	188.333	2.883835
25-35	7.275	134.667	6.022371
95-105	5.000	40.667	3.806608
295-305	7.495	24.000	2.412568

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

121.400	83.710	
83.710	26.400	17.398

****

129.750	87.750	73.250
87.750	73.250	10.420
73.250	10.420	9.965

---

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-16. DTPA-MANGANESE
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	5.450	5.180	2.240	3.140	2.070	3.616	44.420
2.	25-35	2.520	2.580	1.040	0.540	0.660	1.468	68.464
3.	95-105	1.680	1.230	1.290	2.100	1.990	1.658	23.835
4.	295-305	2.090	2.380	2.160	4.360	4.590	3.116	40.047
							2.465	
5.	TOTAL	11.740	11.370	6.730	10.140	9.310	49.290	
6.	MEAN	2.935	2.843	1.683	2.535	2.328		
7.	RSD	58.309	58.681	36.083	95.380	108.117		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	5.315	2.483	-6.432115
25-35	2.550	0.747	-9.208372
95-105	1.455	1.793	0.919582
295-305	2.235	3.703	1.459979

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

3.616 3.116
3.116 1.658 1.468

****

2.935 2.843 2.535 2.328 1.683

---

a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-17. DTPA-NICKEL
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.620	0.730	0.740	2.430	1.300	1.164	64.937
2.	25-35	0.240	0.270	0.540	1.040	0.330	0.484	68.628
3.	95-105	0.200	0.140	0.500	0.390	0.310	0.308	46.904
4.	295-305	0.190	0.240	0.280	0.320	0.440	0.294	32.232
							0.563	
5.	TOTAL	1.250	1.380	2.060	4.180	2.380	11.250	
6.	MEAN	0.313	0.345	0.515	1.045	0.595		
7.	RSD	65.963	76.120	36.620	108.406	83.081		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	0.675	1.490	1.267567
25-35	0.255	0.637	1.402727
95-105	0.170	0.400	3.085774
295-305	0.215	0.347	2.031885

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

1.164
0.484 0.308 0.294

****

1.045 0.595 0.515
0.595 0.515 0.345 0.313

---

a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-18. DTPA-COBALT
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.080	0.040	0.100	0.140	0.040	0.080	53.033
2.	25-35	0.070	0.010	0.070	0.050	0.040	0.048	51.875
3.	95-105	0.010	0.010	0.050	0.030	0.030	0.026	64.358
4.	295-305	0.010	0.010	0.090	0.040	0.050	0.040	82.916
							0.049	
5.	TOTAL	0.170	0.070	0.310	0.260	0.160	0.970	
6.	MEAN	0.043	0.018	0.078	0.065	0.040		
7.	RSD	88.822	85.714	28.611	65.271	96.825		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	0.060	0.093	0.825723
25-35	0.040	0.053	0.531369
95-105	0.010	0.037	3.098387
295-305	0.010	0.060	2.535463

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

0.080	0.048	
0.048	0.040	0.026

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0.078	0.065	0.043	0.040
0.065	0.043	0.040	
0.043	0.040	0.018	

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a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-19. DTPA-ZINC
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.980	1.590	10.560	8.040	7.520	5.738	73.719
2.	25-35	0.120	0.210	3.420	1.100	1.420	1.254	106.383
3.	95-105	0.040	0.030	0.210	0.240	0.270	0.158	72.357
4.	295-305	0.050	0.070	0.230	0.180	0.320	0.170	66.160
							1.830	
5.	TOTAL	1.190	1.900	14.420	9.560	9.530	36.600	
6.	MEAN	0.298	0.475	3.605	2.390	2.383		
7.	RSD	153.408	157.332	135.254	112.969	108.717		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	1.285	8.707	6.018932
25-35	0.165	1.980	1.935515
95-105	0.035	0.240	9.043140
295-305	0.060	0.243	3.433033

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

5.738
1.254 0.170 0.158

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3.605 2.390 2.383 0.475 0.298

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a. Relative standard deviation
b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-20. DTPA-CADMIUM
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	0.080	0.120	0.140	0.240	0.180	0.152	40.126
2.	25-35	0.010	0.020	0.070	0.050	0.040	0.038	62.828
3.	95-105	0.010	0.010	0.020	0.010	0.010	0.012	37.268
4.	295-305	0.010	0.010	0.030	0.010	0.010	0.014	63.888
							0.054	
5.	TOTAL	0.110	0.160	0.260	0.310	0.240	1.080	
6.	MEAN	0.028	0.040	0.065	0.078	0.060		
7.	RSD	127.273	133.853	83.796	98.056	23.570		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	0.100	0.187	2.146879
25-35	0.015	0.053	3.199798
95-105	0.010	0.013	0.774597
295-305	0.010	0.017	0.774597

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

0.152
0.038 0.014 0.012

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0.078 0.065 0.060 0.040 0.028

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a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-21. DTPA-COPPER
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	2.410	4.030	6.720	8.230	6.200	5.518	41.661
2.	25-35	0.350	0.360	2.700	2.230	1.440	1.416	75.431
3.	95-105	0.710	0.170	1.490	0.870	0.500	0.748	65.579
4.	295-305	0.240	0.370	2.290	1.190	0.600	0.938	89.442
							2.155	
5.	TOTAL	3.710	4.930	13.200	12.520	8.740	43.100	
6.	MEAN	0.928	1.233	3.300	3.130	2.185		
7.	RSD	108.734	151.503	70.749	105.472	42.282		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	3.220	7.050	3.864505
25-35	0.355	2.123	3.725866
95-105	0.440	0.953	1.211560
295-305	0.305	1.360	1.645489

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

5.518		
1.416	0.938	0.748

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3.300	3.130	2.185
3.130	2.185	
2.185	1.233	0.928

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a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE B-22. DTPA-LEAD
ppm

		CONTROL SITES		TREATMENT SITES			MEAN	RSD[a]
		(1)	(2)	(3)	(4)	(5)	(6)	(7)
1.	0-16	5.080	10.370	3.520	5.680	6.580	6.246	40.998
2.	25-35	0.170	0.360	0.590	0.670	0.280	0.414	50.799
3.	95-105	0.140	0.090	0.350	0.290	0.170	0.208	52.048
4.	295-305	0.120	0.170	0.410	0.360	0.170	0.246	52.736
							1.779	
5.	TOTAL	5.510	10.990	4.870	7.000	7.200	35.570	
6.	MEAN	1.378	2.748	1.218	1.750	1.800		
7.	RSD	179.196	185.002	126.356	121.753	312.206		

	CONTROL SITE AVERAGE	TREATMENT SITE AVERAGE	"t"
0-16	7.725	5.260	-1.074722
25-35	0.265	0.513	1.468735
95-105	0.115	0.270	2.188993
295-305	0.145	0.313	1.749809

GROUPS OF NONHETEROGENEOUS MEANS.
DUNCAN'S MULTIPLE RANGE TEST.[b]

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6.246
0.414      0.246      0.208

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2.748      1.800      1.750      1.378      1.218
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a. Relative standard deviation

b. Numbers on same horizontal line are nonheterogenous. First set of numbers refers to Mean column (6); second set refers to Mean line 6.

TABLE C-1. WELL 3A GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	1976 July	1976 Sept	1976 Dec	Mar 1977 1	Mar 1977 2	Average	Standard deviation	Coefficient of variation
COD	60	27	71	39	32	46	19	0.41
BOD	3	<6	16	5	<2	6	5.6	0.93
TOC	--	--	--	10	--	--	--	--
CCE	--	--	--	2.2	--	--	--	--
Total N	4.1	3.5	2.7	3.7	4.9	3.8	0.8	0.21
NH_3-N	<0.2	1.6	0.1	0	--	<0.4	<0.8	2.0
N org	3.8	1.0	0.9	1.1	4.3	2.2	1.7	0.77
NO_3-N	0.14	0.9	1.8	2.6	0.6	1.2	1.0	0.83
Total P	6.5	7.9	11.4	7.4	6.6	8.0	2.0	0.25
PO_4-P	5.6	5.8	9.4	7.4	--	6.8	1.8	0.26
Total coliform, count/100 ml	$>2.4 \times 10^5$	4.6×10^6	35,000	350,000	240,000	1.1×10^6	2.0×10^6	1.82
Fecal coliform, count/100 ml	$>2.4 \times 10^5$	430,000	35,000	24,000	49,000	156,000	177,000	1.13
pH, units	7.5	7.4	7.7	7.5	--	7.5	0.1	0.01
TDS	1,285	1,504	1,200	1,138	--	1,282	160	0.12
Conductivity, umhos/cm	1,620	2,220	1,580	1,420	--	1,710	351	0.21
SS	--	--	--	--	15	--	--	--
Alkalinity	--	--	521	514	580	538	36	0.07
B	1.0	1.7	1.2	1.0	--	1.2	0.3	0.25
F	1.5	1.4	0.4	0.9	--	1.1	0.5	0.45
Na	230	327	239	219	230	249	44	0.18
Ca	119	106	82	68	126	100	25	0.25
Mg	57	78	78	74	70	71	9.0	0.13
K	12.5	16.0	12.1	11.3	12.9	13.0	1.8	0.14
Cl	258	402	250	215	--	281	83	0.3
SO_4	65	228	194	168	--	164	70	0.43
Ag	0.006	<0.005	0.005	<0.005	--	<0.006	<0.001	0.17
As	<0.01	<0.01	0.01	<0.01	--	<0.01	--	--
Ba	<0.1	<0.1	<0.02	0.23	--	<0.11	<0.09	0.82
Cd	0.003	0.010	0.010	0.003	--	0.007	0.004	0.57
Co	0.008	0.012	<0.006	<0.006	--	<0.008	<0.003	0.38
Cr	<0.004	<0.004	<0.004	0.032	--	<0.011	<0.014	1.27
Cu	0.030	0.054	0.038	0.04	--	0.041	0.010	0.24
Fe	0.49	0.12	<0.01	0.20	--	0.21	0.21	1.0
Hg	<0.001	<0.001	<0.001	<0.001	--	<0.001	--	--
Mn	1.65	0.81	1.8	0.94	--	1.30	0.50	0.38
Ni	0.075	0.115	0.04	0.16	--	0.10	0.05	0.5
Pb	0.009	0.10	0.11	0.06	--	0.10	0.06	0.6
Se	<0.001	<0.001	<0.001	<0.001	--	<0.001	--	--
Zn	0.066	0.084	0.021	0.13	--	0.073	0.041	0.56
SAR	4.34	5.88	4.53	4.37	4.07	4.64	0.71	0.15

TABLE C-2. WELL 5A GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

	1976			Mar 1977				
Constituent	July	Sep.	Dec	1	2	Average	Standard deviation	Coefficient of variation
COD	--	56	78	33	32	50	22	0.44
BOD	10	<6	43	5	2	13	17	1.3
TOC	--	--	--	11	--	--	--	--
CCE	--	--	--	1.7	--	--	--	--
Total N	<0.2	1.4	2.8	2.8	1.4	1.7	1.1	0.65
NH_3-N	<0.2	<0.1	<0.1	0.5	--	<0.2	<0.2	1.0
N org	<0.2	<1.3	1.0	1.4	1.3	<1.0	<0.5	0.5
NO_3-N	0.05	0.11	1.8	0.9	0.1	0.6	0.8	1.3
Total P	0.13	19	11.2	10.2	7.3	9.6	6.8	0.71
PO_4-P	0.13	13	9.9	10.2	--	8.3	5.6	0.67
Total coliform, count/100 ml	54,000	1.1×10^6	<2	1,600	1,300	231,000	486,000	2.1
Fecal coliform, count/100 ml	<2	930,000	0	1,600	790	186,000	416,000	2.2
pH, units	7.9	7.6	7.5	7.7	--	7.7	0.2	0.03
TDS	1,090	1,462	1,388	1,160	--	1,275	178	0.14
Conductivity, μmhos/cm	1,810	2,020	2,000	1,480	--	1,828	250	0.14
SS	--	--	--	--	10	--	--	--
Alkalinity	--	--	516	519	530	522	7	0.01
B	0.7	1.7	1.3	1.1	--	1.2	0.4	0.33
F	0.75	--	0.4	1.2	--	0.80	0.4	0.5
Na	248	308	285	223	242	261	34	0.13
Ca	133	101	120	71	112	107	23	0.21
Mg	75	70	85	66	--	74	7	0.09
K	13.6	16.4	17.6	13.3	--	14.9	2.0	0.13
Cl	280	342	315	230	--	292	48	0.16
SO_4	280	280	218	210	--	247	38	0.15
Ag	0.008	<0.005	0.008	<0.005	--	<0.007	<0.002	0.29
As	<0.01	<0.01	<0.01	<0.01	--	<0.01	--	--
Ba	<0.1	<0.1	<0.02	0.38	--	<0.15	<0.16	<1.1
Cd	0.005	0.085	0.10	0.008	--	0.050	0.050	1.0
Co	0.018	0.019	<0.006	0.006	--	<0.012	<0.007	0.58
Cr	<0.004	<0.004	<0.004	0.055	--	<0.017	<0.026	1.53
Cu	0.014	0.058	0.052	0.02	--	0.036	0.022	0.61
Fe	0.55	0.19	0.04	1.25	--	0.51	0.54	1.1
Hg	<0.001	<0.001	<0.001	<0.001	--	<0.001	--	--
Mn	0.70	0.81	0.8	0.18	--	0.62	0.30	0.48
Ni	0.115	0.115	0.04	0.38	--	0.16	0.15	0.94
Pb	<0.005	0.10	0.16	0.04	--	<0.08	<0.07	0.88
Se	<0.001	<0.001	<0.001	<0.001	--	<0.001	--	--
Zn	0.118	0.073	0.037	0.13	--	0.090	0.043	0.48
SAR	4.26	5.76	4.86	4.44	4.48	4.76	0.60	0.13

TABLE C-3. WELL 1B GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

	1976						
Constituent	July	Sept	Dec	Mar 1977	Average	Standard deviation	Coefficient of variation
COD	25	10	32	18	21	9	0.43
BOD	3	0	5	<2	<3	<2	0.85
TOC	--	--	--	9	9	--	--
CCE	0.72	--	--	--	0.72	--	--
Total N	7.1	4.0	5.4	10.1	6.7	2.6	0.39
NH_3-N	<0.2	<0.1	0.4	0.0	<0.2	<0.2	1.0
N org	0.2	<0.1	0.6	1.1	<0.5	<0.45	0.9
NO_3-N	6.9	4.0	4.4	9.0	6.1	2.3	0.38
Total P	0.06	0.13	0.08	0.3	0.14	0.11	0.79
PO_4-P	0.06	0.11	0.04	0.3	0.13	0.12	0.92
Total coliform, count/100 ml	<2	<2	<2	<2	<2	--	--
Fecal coliform, count/100 ml	0	0	0	0	0	--	--
pH, units	8.1	7.8	8.7	8.3	8.2	0.4	0.05
TDS	846	817	748	804	804	41	0.05
Conductivity, µmhos/cm	1,170	1,210	1,090	1,000	1,118	93	0.08
SS	--	--	--	--	--	--	--
Alkalinity	--	--	341	470	406	91	0.22
B	0.5	0.7	0.7	0.8	0.7	0.1	0.14
F	0.4	0.4	0.2	0.4	0.35	0.1	0.29
Na	115	143	133	166	139	21	0.15
Ca	57	30	57	38	46	14	0.30
Mg	77	76	84	87	81	5	0.06
K	3.5	3.5	4.7	3.9	3.9	0.6	0.15
Cl	88	82	60	92	80	14	0.18
SO_4	210	228	200	186	206	18	0.09
Ag	<0.004	<0.005	0.003	<0.005	<0.004	<0.001	0.25
As	<0.01	<0.01	<0.01	<0.01	<0.01	--	--
Ba	<0.1	<0.1	0.08	0.16	<0.11	<0.04	0.36
Cd	0.002	<0.001	0.008	0.004	0.004	0.003	0.75
Co	0.010	<0.006	<0.006	<0.006	<0.007	<0.002	0.29
Cr	<0.004	<0.004	0.01	<0.004	<0.006	<0.003	0.5
Cu	0.04	0.039	0.09	0.07	0.06	0.02	0.33
Fe	1.25	0.14	0.07	0.38	0.46	0.54	1.17
Hg	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Mn	0.40	0.083	0.22	0.18	0.22	0.13	0.59
Ni	0.04	0.055	0.02	0.028	0.04	0.02	0.5
Pb	0.04	0.015	0.085	0.06	0.05	0.03	0.6
Se	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Zn	0.16	0.14	0.70	0.14	0.29	0.28	0.97
SAR	2.33	3.16	2.62	3.39	2.88	0.49	0.17

TABLE C-4. WELL 3B GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	1976 July	1976 Sept	1976 Dec	Mar 1977 1	Mar 1977 2	Average	Standard deviation	Coefficient of variation
COD	22	19	24	46	8	24	4	0.17
BOD	<6	3.0	4.8	<2	<2	<4	<1.8	0.45
TOC	--	--	--	<1	--	--	--	--
CCE	--	--	--	--	--	--	--	--
Total N	1.3	3.3	10.7	14.5	9.2	7.8	5.4	0.69
NH_3-N	<0.2	<0.1	<0.1	0	--	<0.1	<0.08	0.8
N org	1.2	1.0	0.8	0.5	0.4	0.8	0.3	0.38
NO_3-N	<0.05	2.3	9.9	14.0	8.8	7.0	5.7	0.81
Total P	--	0.14	0.3	0.2	0.3	0.2	0.08	0.4
PO_4-P	--	0.09	0.01	0.2	--	0.1	0.10	1.0
Total coliform, count/100 ml	<2	2,400	5,400	33	<2	1,570	2,380	1.5
Fecal coliform, count/100 ml	<2	43	8	<2	<2	<12	<18	1.6
pH, units	8.0	7.6	7.6	7.3	--	7.6	0.3	0.04
TDS	1,402	1,331	1,448	1,420	--	1,400	50	0.04
Conductivity, µmhos/cm	1,810	2,000	2,150	1,730	--	1,922	189	0.10
SS	--	--	--	--	18	--	--	--
Alkalinity	--	--	569	622	580	590	28	0.05
B	1.1	1.4	1.4	1.5	--	1.4	0.2	0.14
F	0.3	0.3	0.2	0.4	--	0.3	0.1	0.33
Na	212	271	262	244	242	246	23	0.09
Ca	176	8	128	76	132	104	64	0.62
Mg	94	79	102	99	82	91	10	0.11
K	6.2	9.4	10.5	10.5	10.1	9.3	1.8	0.19
Cl	260	286	315	318	--	295	27	0.09
SO_4	186	210	200	168	--	181	37	0.2
Ag	0.006	<0.005	0.008	<0.005	--	<0.006	<0.001	0.17
As	<0.01	<0.01	<0.01	<0.01	--	<0.01	--	--
Ba	<0.1	<0.1	<0.02	0.26	--	<0.12	<0.10	0.83
Cd	0.005	0.003	0.015	0.003	--	0.007	0.006	0.86
Co	0.035	0.022	<0.006	<0.006	--	<0.017	<0.012	0.71
Cr	<0.004	<0.004	0.02	0.025	--	0.013	0.011	0.85
Cu	0.011	0.047	0.10	0.03	--	0.047	0.038	0.81
Fe	0.43	2.30	0.50	0.20	--	0.86	0.97	1.13
Hg	<0.001	<0.001	<0.001	<0.001	--	<0.001	--	--
Mn	--	0.22	0.75	0.65	--	0.54	0.28	0.52
Ni	0.115	0.138	0.075	0.074	--	0.101	0.032	0.32
Pb	<0.005	0.15	0.10	0.08	--	0.08	0.05	0.75
Se	<0.001	<0.001	<0.001	<0.001	--	<0.001	--	--
Zn	0.13	0.06	0.43	0.13	--	0.19	0.16	0.84
SAR	3.21	6.35	4.19	4.34	4.07	4.43	1.16	0.26

TABLE C-5. WELL 1C GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	1976			Mar 1977	Average	Standard deviation	Coefficient of variation
	July	Sept	Dec				
COD	11	10	32	18	18	10	0.56
BOD	<6.0	3.0	5.5	5.0	<4.9	<1.3	0.27
TOC	--	--	--	13	13	--	--
CCE	0.54	0.78	--	--	0.66	0.17	0.26
Total N	7.9	1.1	1.2	1.2	2.9	3.4	1.17
NH_3-N	<0.2	0.5	0.9	0.0	<0.4	<0.4	1.0
N org	7.9	0.5	0.2	1.1	2.4	3.7	1.5
NO_3-N	<0.05	0.15	0.1	0.1	0.1	0.04	0.4
Total P	<0.03	0.27	0.12	0.06	0.12	0.11	0.92
PO_4-P	<0.03	0.22	0.05	<0.02	<0.08	<0.09	1.1
Total coliform, count/100 ml	<2	23	8	<2	<9	<10	1.1
Fecal coliform, count/100 ml	<2	<2	<2	<2	<2	0	1.0
pH, units	8.1	7.7	7.9	8.4	8.0	0.3	0.38
TDS	1,277	1,376	1,266	1,334	1,313	51	0.04
Conductivity, µmhos/cm	1,700	1,890	1,800	1,500	1,723	167	0.10
SS	--	--	--	--	--	--	--
Alkalinity	--	--	485	553	519	48	0.09
B	0.5	0.2	0.6	0.4	0.4	0.2	0.5
F	0.4	0.3	0.2	0.4	0.3	0.1	0.33
Na	214	207	212	193	207	9	0.04
Ca	69	26	77	50	56	23	0.95
Mg	115	109	112	135	118	12	0.10
K	4.7	5.1	5.9	5.1	5.2	0.5	0.10
Cl	136	144	110	143	133	16	0.12
SO_4	375	400	398	295	367	49	0.13
Ag	0.006	<0.005	0.032	0.008	<0.013	<0.013	1.0
As	<0.01	<0.01	<0.01	<0.01	<0.01	--	--
Ba	<0.1	<0.1	0.08	0.15	<0.11	<0.03	0.27
Cd	0.004	<0.001	0.008	0.002	<0.004	<0.003	0.75
Co	0.012	0.009	<0.006	<0.006	<0.008	<0.003	0.38
Cr	<0.004	<0.004	<0.004	0.008	<0.005	<0.002	0.40
Cu	0.009	0.066	0.10	<0.004	0.045	0.046	1.0
Fe	0.030	0.010	0.08	<0.04	<0.04	<0.029	0.73
Hg	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Mn	0.360	0.18	0.14	0.15	0.21	0.10	0.48
Ni	0.080	0.080	0.02	0.010	0.048	0.038	0.79
Pb	0.011	0.010	0.055	0.10	0.044	0.043	0.98
Se	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Zn	0.110	0.138	0.70	0.14	0.27	0.29	1.1
SAR	3.66	3.97	3.61	3.22	3.61	0.31	0.09

TABLE C-6. WELL 2C GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	1976 July	1976 Sept	1976 Dec	Mar 1977	Average	Standard deviation	Coefficient of variation
COD	140	199	86	81	127	55	0.43
BOD	<6	46	6	5	<16	<20	1.25
TOC	--	--	--	20	20	--	--
CCE	--	--	--	--	--	--	--
Total N	0.4	1.6	2.9	2.1	1.8	1.1	0.61
NH_3-N	<0.2	<0.1	<0.1	0.7	<0.3	<0.3	1.0
N org	0.3	1.6	1.5	1.0	1.1	0.6	0.55
NO_3-N	0.08	<0.05	1.4	0.4	0.5	0.6	1.2
Total P	0.06	0.07	0.14	0.06	0.08	0.04	0.5
PO_4-P	0.06	<0.02	0.02	<0.02	<0.03	<0.02	0.67
Total coliform, count/100 ml	540	4,600	3,500	<2	2,161	2,238	1.0
Fecal coliform, count/100	<2	1,100	<2	<2	276	549	1.99
pH, units	8.0	7.6	8.0	7.9	7.9	0.2	0.03
TDS	830	790	718	702	760	60	0.08
Conductivity, µmhos/cm	1,210	1,200	1,110	907	1,107	141	0.13
SS	--	--	--	--	--	--	--
Alkalinity	--	--	440	420	430	14	0.03
B	0.3	0.8	0.6	0.5	0.6	0.2	0.33
F	0.4	0.3	0.2	0.2	0.3	0.1	0.33
Na	120	129	133	156	135	15	0.11
Ca	63	240	47	30	95	98	1.0
Mg	79	75	80	68	76	5	0.07
K	5.5	5.1	5.9	5.5	5.5	0.3	0.05
Cl	94	50	70	109	81	26	0.32
SO_4	186	180	103	108	144	45	0.31
Ag	<0.004	<0.005	0.007	<0.005	<0.006	<0.002	0.33
As	<0.01	<0.01	<0.01	<0.01	<0.01	--	--
Ba	<0.1	<0.1	0.09	0.15	<0.11	<0.03	0.27
Cd	0.003	<0.001	0.014	0.004	0.006	0.006	1.0
Co	<0.006	0.008	<0.006	<0.006	<0.007	<0.001	0.14
Cr	<0.004	<0.004	0.08	<0.004	<0.03	<0.040	1.33
Cu	0.032	0.008	0.12	0.09	0.063	0.052	0.83
Fe	0.30	0.007	0.50	<0.04	0.21	0.23	1.1
Hg	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Mn	0.130	0.068	0.084	0.12	0.101	0.029	0.29
Ni	0.055	0.060	0.015	0.024	0.039	0.022	0.56
Pb	0.010	0.036	0.07	0.15	0.067	0.061	0.91
Se	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Zn	0.137	0.047	0.041	0.16	0.10	0.06	0.6
SAR	2.38	1.86	2.74	3.60	2.65	0.73	0.28

TABLE C-7. WELL 4C GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	1976 July	1976 Sept	1976 Dec	Mar 1977	Average	Standard deviation	Coefficient of variation
COD	10	10	28	15	16	9	0.56
BOD	2	2	4	<2	<2.5	<1	0.4
TOC	--	--	--	<1	--	--	--
CCE	--	--	0.65	--	--	--	--
Total N	1.7	1.6	2.1	2.9	2.1	0.6	0.29
NH_3-N	<0.2	<0.1	<0.1	0.0	<0.1	<0.08	0.73
N org	<0.2	<0.1	0.5	<1.0	<0.5	<0.4	0.8
NO_3-N	1.7	1.6	1.6	2.9	1.9	0.6	0.32
Total P	0.05	0.06	0.07	0.08	0.07	0.01	0.14
PO_4-P	0.05	0.03	0.01	0.06	0.04	0.02	0.5
Total coliform, count/100 ml	<2	<2	<2	<2	<2	--	--
Fecal coliform, count/100 ml	<2	<2	<2	<2	<2	--	--
pH, units	8.0	7.7	7.7	7.8	7.8	0.1	0.01
TDS	855	857	740	826	820	55	0.07
Conductivity, μmhos/cm	1,190	1,230	1,170	1,000	1,148	101	0.09
SS	--	--	--	--	--	--	--
Alkalinity	--	--	411	439	425	20	0.05
B	0.5	0.8	0.7	0.7	0.7	0.1	0.14
F	0.5	0.5	0.4	0.2	0.4	0.1	0.25
Na	101	110	106	129	112	12	0.11
Ca	61	27	65	34	47	19	0.40
Mg	87	83	95	89	89	5	0.06
K	3.5	3.5	4.7	4.7	4.1	0.7	0.17
Cl	88	82	50	82	76	17	0.22
SO_4	194	193	180	168	184	12	0.07
Ag	<0.004	<0.005	0.004	<0.005	<0.005	<0.001	0.2
As	<0.01	<0.01	<0.01	<0.01	<0.01	--	--
Ba	<0.1	<0.1	0.03	<0.02	<0.07	<0.05	0.71
Cd	0.003	<0.001	0.007	<0.001	<0.003	<0.003	1.0
Co	<0.006	<0.006	<0.006	<0.006	<0.006	--	--
Cr	<0.004	<0.004	0.09	0.030	<0.032	<0.041	1.28
Cu	0.012	0.04	0.042	<0.004	0.025	0.019	0.76
Fe	0.030	<0.04	<0.01	<0.04	<0.03	<0.014	0.47
Hg	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Mn	<0.003	0.007	0.003	0.005	<0.005	<0.002	0.40
Ni	0.050	0.055	0.045	<0.005	0.039	0.023	0.59
Pb	0.008	0.008	0.012	0.02	0.012	0.006	0.5
Se	<0.001	<0.001	<0.001	<0.001	<0.001	--	--
Zn	0.050	0.068	0.041	0.04	0.05	0.01	0.2
SAR	1.94	2.37	1.96	2.64	2.23	0.34	0.15

TABLE C-8. WELL 6C GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	July 1976*
COD	14
BOD	6
TOC	--
CCE	--
Total N	<0.2
NH_3-N	<0.2
N org	<0.2
NO_3-N	<0.05
Total P	0.04
PO_4-P	0.04
Total coliform, count/100 ml	<2
Fecal coliform, count/100 ml	<2
pH, units	7.2
TDS	855
Conductivity, µmhos/cm	1,950
SS	--
Alkalinity	--
B	1.4
F	0.3
Na	230
Ca	217
Mg	103
K	4.5
Cl	250
SO_4	280
Ag	0.008
As	<0.01
Ba	<0.1
Cd	0.003
Co	0.008
Cr	<0.004
Cu	0.008
Fe	0.020
Hg	<0.001
Mn	<0.003
Ni	0.070
Pb	<0.005
Se	<0.001
Zn	0.002
SAR	3.22

*Only date sampled.

TABLE C-9. WELL 7C GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	1976 July	1976 Sept	1976 Dec	Mar 1977	Average	Standard deviation	Coefficient of variation
COD	<10	<10	--	22	<14	<7	0.5
BOD	<6	<6	--	3	<5	<2	0.42
TOC	--	--	--	6	--	--	--
CCE	--	--	--	--	--	--	--
Total N	<0.2	5.6	--	12.2	<6.0	<6.0	1.0
NH_3-N	<0.2	<0.2	--	0.0	<0.2	<0.2	1.0
N org	<0.2	<0.2	--	1.2	<0.5	<0.6	1.2
NO_3-N	<0.05	5.6	--	11.0	5.6	5.5	0.98
Total P	0.04	0.01	--	0.05	0.03	0.02	0.67
PO_4-P	0.04	<0.01	--	<0.02	<0.02	<0.02	1.0
Total coliform, count/100 ml	<2	<2	--	23	<9	<12	1.33
Fecal coliform, count/100 ml	<2	<2	--	<2	<2	--	--
pH, units	7.5	7.8	--	7.8	7.7	0.2	0.03
TDS	951	962	--	1,260	1,058	175	0.17
Conductivity, μmhos/cm	1,400	1,590	--	1,580	1,523	107	0.07
SS	--	--	--	--	--	--	--
Alkalinity	--	--	--	556	--	--	--
B	1.1	0.8	--	1.0	1.0	0.2	0.2
F	0.6	0.6	--	0.4	0.5	0.1	0.2
Na	120	147	--	219	162	51	0.31
Ca	62	230	--	54	115	99	0.86
Mg	85	91	--	108	95	12	0.13
K	2.7	2.7	--	4.7	3.4	1.2	0.35
Cl	138	156	--	284	193	80	0.41
SO_4	280	200	--	194	225	48	0.21
Ag	<0.004	<0.005	--	<0.005	<0.005	<0.001	0.2
As	<0.01	<0.01	--	<0.01	<0.01	--	--
Ba	<0.1	<0.1	--	0.20	<0.13	<0.06	0.46
Cd	0.002	0.002	--	0.06	0.02	0.03	1.5
Co	<0.006	<0.006	--	<0.006	<0.006	--	--
Cr	<0.004	<0.004	--	<0.004	<0.004	--	--
Cu	0.006	<0.004	--	<0.004	<0.005	<0.002	0.4
Fe	0.125	<0.04	--	0.24	0.14	0.10	0.71
Hg	<0.001	<0.001	--	<0.001	<0.001	--	--
Mn	0.003	<0.003	--	0.015	0.007	0.007	1.0
Ni	0.055	<0.005	--	0.022	0.027	0.025	0.93
Pb	<0.005	<0.005	--	0.02	<0.010	<0.009	0.9
Se	<0.001	<0.001	--	<0.001	<0.001	--	--
Zn	0.002	0.003	--	0.06	0.02	0.03	1.5
SAR	2.32	2.08	--	3.96	2.71	1.02	0.38

TABLE C-10. WELL 8C GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

Constituent	1976 July	 Sept	 Dec	Mar 1977	Average	Standard deviation	Coefficient of variation
COD	--	18	24	--	21	4	0.19
BOD	--	<6	2	--	<4	<3	0.82
TOC	--	--	--	--	--	--	--
CCE	--	--	--	--	--	--	--
Total N	--	<0.2	7.9	--	4.1	5.4	1.3
NH_3-N	--	<0.2	<0.1	--	<0.2	<0.1	0.5
N org	--	<0.2	0.2	--	<0.2	--	--
NO_3-N	--	<0.1	7.7	--	<3.9	<5.4	1.38
Total P	--	0.1	0.18	--	0.14	0.06	0.43
PO_4-P	--	0.1	0.01	--	0.06	0.06	1.0
Total coliform, count/100 ml	--	<2	130	--	66	91	1.4
Fecal coliform, count/100 ml	--	< 2	<2	--	<2	--	--
pH, units	--	7.8	7.4	--	7.6	0.3	0.04
TDS	--	1,179	1,348	--	1,264	120	0.09
Conductivity, µmhos/cm	--	2,010	2,230	--	2,120	156	0.07
SS	--	--	--	--	--	--	--
Alkalinity	--	--	570	--	--	--	--
B	--	0.9	1.3	--	1.1	0.3	0.27
F	--	0.5	0.3	--	0.4	0.1	0.25
Na	--	285	239	--	262	33	0.13
Ca	--	170	130	--	150	28	0.19
Mg	--	84	103	--	94	13	0.14
K	--	5.3	5.1	--	5.2	0.1	0.02
Cl	--	298	265	--	282	23	0.08
SO_4	--	210	180	--	195	21	0.11
Ag	--	<0.005	0.006	--	<0.006	<0.001	0.17
As	--	<0.01	<0.01	--	<0.01	--	--
Ba	--	<0.1	<0.02	--	<0.06	<0.06	1.0
Cd	--	0.006	0.007	--	0.007	0.001	0.14
Co	--	<0.006	<0.006	--	<0.006	--	--
Cr	--	<0.004	<0.004	--	<0.004	--	--
Cu	--	0.007	0.02	--	0.01	0.01	1.0
Fe	--	0.14	<0.01	--	0.08	0.09	1.1
Hg	--	<0.001	<0.001	--	<0.001	--	--
Mn	--	<0.003	0.009	--	<0.006	<0.004	0.67
Ni	--	<0.005	0.09	--	<0.05	0.06	1.21
Pb	--	<0.005	<0.005	--	<0.005	--	--
Se	--	<0.001	<0.001	--	<0.001	--	--
Zn	--	0.005	0.012	--	0.009	0.005	0.56
SAR	--	4.47	3.80	--	4.13	0.47	0.11

TABLE C-11. WELL 9C GROUNDWATER QUALITY RESULTS
mg/L Unless Otherwise Noted

	1976						
Constituent	July	Sept	Dec	Mar 1977	Average	Standard deviation	Coefficient of variation
COD	--	--	52	<8	30	31	1.0
BOD	--	--	0.3	<2	1.2	1.2	1.0
TOC	--	--	--	5	--	--	--
CCE	--	--	0.71	--	0.71	--	--
Total N	--	--	10.2	9.5	9.9	0.5	0.05
NH_3-N	--	--	<0.1	0	<0.05	--	--
N org	--	--	0.3	0.4	0.4	0.07	0.18
NO_3-N	--	--	9.9	9.1	9.5	0.6	0.06
Total P	--	--	0.13	<0.02	<0.08	<0.08	1.0
PO_4-P	--	--	0.0	<0.02	<0.01	<0.01	--
Total coliform, count/100 ml	--	--	33	540	287	359	1.3
Fecal coliform, count/100 ml	--	--	8	540	274	376	14
pH, units	--	--	8.3	7.5	7.9	0.6	0.08
TDS	--	--	1,274	1,312	1,293	27	0.02
Conductivity, µmhos/cm	--	--	1,940	1,510	1,725	304	0.18
SS	--	--	--	--	--	--	--
Alkalinity	--	--	579	568	574	7.8	0.01
B	--	--	1.1	1.0	1.1	0.07	0.06
F	--	--	0.25	0.4	0.3	0.1	0.33
Na	--	--	230	219	225	8	0.04
Ca	--	--	100	59	80	29	0.36
Mg	--	--	107	111	109	3	0.03
K	--	--	5.5	5.1	5.3	0.3	0.06
Cl	--	--	265	267	266	1	0.004
SO_4	--	--	16	194	105	126	0.83
Ag	--	--	0.005	<0.005	<0.005	--	--
As	--	--	<0.01	<0.01	<0.01	--	--
Ba	--	--	<0.02	0.06	<0.04	<0.03	0.75
Cd	--	--	0.007	<0.001	<0.004	<0.005	1.3
Co	--	--	<0.006	<0.006	<0.006	--	--
Cr	--	--	<0.004	<0.004	<0.004	--	--
Cu	--	--	0.032	<0.004	<0.018	<0.020	1.1
Fe	--	--	<0.01	<0.04	<0.03	<0.03	1.0
Hg	--	--	<0.001	<0.001	<0.001	--	--
Mn	--	--	<0.003	0.018	<0.011	<0.011	1.0
Ni	--	--	0.035	0.016	0.026	0.013	0.5
Pb	--	--	<0.05	0.015	0.033	0.025	0.76
Se	--	--	<0.001	<0.001	<0.001	--	--
Zn	--	--	0.076	0.004	0.04	0.051	1.3
SAR	--	--	3.81	3.88	3.84	0.05	0.01